Forest–Fish Conference:

Land Management Practices Affecting Aquatic Ecosystems

M.K. Brewin and D.M.A. Monita
Technical Coordinators

Information Report NOR-X-356

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ERRATUM

Page 273. Figure 7 caption: For "Fredle Indices from both top and bottom layers of freeze-cores obtained from Kynock, Forfar and Gluskie creeks, 1990 to" read "Fredle Indices from both top and bottom layers of freeze-cores obtained from Kynock, Forfar and Gluskie creeks, 1990 to 1994."

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May 1–4, 1996

M. K. Brewin and D.M.A. Monita, technical coordinators

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Abstract

The Forest-Fish Conference was held on May 1-4, 1996, in Calgary Alberta. The conference provided a forum for the exchange of information concerning the relationships between forest land-use activities and aquatic resources among an international assemblage of technical experts. Presentations were made on the relationships between forest harvesting activities and stream flows; road construction and sedimentation; degraded riparian area recovery and improved livestock management; stream ecosystem protection; and timber harvest and riparian buffer requirements. Management solutions that improve watershed protection and minimize the impacts of forest land-use activities on aquatic environments were discussed. Fifty papers from this meeting are presented in this volume.

Résumé

La conférence concernant la forêt et les ressources halieutiques s'est déroulée du 1er au 4 mai 1996 à Calgary, en Alberta. Elle a permis à nombre d'experts techniques de partout au monde d'échanger de l'information sur les liens entre les activités d'utilisation des terrains forestiers et les ressources aquatiques. Les participants ont présenté des exposés sur les liens entre les activités d'exploitation forestière et l'écoulement fluviatile; la construction routière et la sédimentation; la remise en état des zones riveraines dégradées et l'amélioration de la gestion du bétail; l'exploitation forestière et les exigences en matière de zone tampon riveraine; ainsi que sur la protection des écosystèmes aquatiques. Ils ont également traité des solutions de gestion qui améliorent la protection des bassins versants et minimisent les impacts des activités d'utilisation des terrains forestiers sur les milieux aquatiques. Cinquante exposés présentés lors de cette conférence figurent dans ce recueil.

Contents

NOTE

The views, conclusions, and recommendations published in this proceedings are those of the authors and do not necessarily imply endorsement by the Canadian Forest Service.

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Preface

Like many other natural resources, pure, clean water supplies are often taken for granted, particularly in western North America. However, the general public's concern about the potential impacts that various land-use management practices have on aquatic resources is increasing. This growing concern is driving the need for a better understanding of how forest management practices affect such things as stream flow regimes, water quantity and quality, and fish habitat. Of equal, if not greater, importance is the development and implementation of workable management solutions.

The recognition of these needs, and concerns from scientists and resource managers with their decreasing ability to keep up with the current literature, led to a series of discussions between stakeholder groups in Alberta. These discussions resulted in the creation of a multi-stakeholder Steering Committee to organize an international, scientific conference focusing on the relationships between forest land-use activities and aquatic resources. The Steering Committee included representation from conservation organizations, government agencies, academia, the forest industry, and the oil and gas industry. There was unanimous agreement among the Committee that the Conference should not just focus on the problems, but it should also concentrate on potential management solutions. Consequently, the "Forest–Fish Conference: Land Management Practices Affecting Aquatic Ecosystems" was organized to: 1) facilitate the exchange of information concerning the relationships between forest land-use activities and aquatic resources among an international assemblage of technical experts; 2) increase awareness of management solutions that improve watershed protection and minimize the impacts of forest land-use activities on aquatic environments; 3) increase opportunities for stakeholders to work together to find cooperative solutions to potential forestry-related problems; and 4) identify benefits, or ways and means of providing benefits, of forest management practices affecting aquatic resources.

The Forest–Fish Conference was held in Calgary, Alberta from May 1 to 4, 1996. It resulted in approximately 60 oral and 15 poster presentations on the relationships between forest harvest and stream flows, road construction and sedimentation, degraded riparian area recovery and improved livestock management, stream ecosystem protection, and timber harvest and riparian buffer requirements. It was attended by 230 delegates from four countries (Canada, the United States, Sweden and Finland), which included representation from seven Canadian provinces, the Yukon Territories and eight U.S. states.

We appreciate the dedicated efforts of the authors for their contributions towards what we believe will become a valuable reference document that helps advance a better understanding of the relationships between forest management practices and aquatic resources. We trust readers of these proceedings will share this belief.

M.K. Brewin

D.M.A. Monita

Technical Coordinators

Acknowledgments

In addition to the members of the multi-agency Steering Committee that helped coordinate the Forest–Fish Conference, there are many other individuals who contributed to the success of the conference and these proceedings. The Steering Committee gratefully acknowledges the financial and in-kind contributions of the sponsors, which are listed below. We thank and acknowledge the National Research Council Press for permission to reproduce the paper entitled, "Hydrology of brook trout *(Salvelinus fontinalis)* spawning and incubation habitats:implications for forestry and land use development", which originally appeared in the Canadian Journal of Forest Research. Brenda Laishley, Elaine Schiewe, and Anne Cardasilaris, of the Canadian Forest Service, are acknowledged for overseeing the editorial and production stages of the proceedings. Lu Ziola, of some production!, provided graphic support and typesetting services. There were also numerous anonymous reviewers whose comments of draft manuscripts helped contribute to the quaility of these proceedings. Support from Trout Unlimited Canada staff (Nik Homberg, Cathy Nelson, Mike Monita and Patti Brewin) who provided much of administrative and logistical support associated with hosting the conference was also gratefully appreciated. We also thank several of the volunteers who assisted us with some of the behind-the-scenes work during the conference: Sandra Foss, Jim Rouse, Tony Blake, Henry Komadowski, Barb Ainsley, Liane Spillios, James Dunnigan, Jim McCarthy, and Kelly Chambers. A special thanks is reserved for speakers who gave keynote and luncheon presentations: Dr. Stanley Gregory, Department of Fisheries and Wildlife, Oregon State University, Corvallis, Oregon; Don Oman, Twin Falls Ranger District, Sawtooth National Forest, Twin City Falls, Idaho; and Dr. Morley Barrett, Natural Resources Service, Alberta Environmental Protection, Edmonton, Alberta.

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Abstract

We summarize the results of 5 years of research on instream habitat structures in southwestern Alberta, and provide information on the efficiency of these fish habitat improvement devices. The short-term performance of 351 instream structures was investigated in southwestern Alberta during 1991 and 1992. The majority of these structures were between 2 and 7 years of age (90% of sample) and none had been subjected to a flood >6 year return period. Under these conditions, 63% of the structures were found to have maintained their physical stability, or had only minor faults. Sixty-one percent of the surveyed structures provided the desired deepwater refuge fish habitat. This information was re-analyzed between 1993 and 1994 to determine relationships between structure performance and fluvial and hydraulic characteristics. This investigation concluded that structures tended to perform better in stable channels with low rates of bedload transport. Following a sizeable flood in June 1995 (>100 year return period), a subset of the original structures was re-evaluated. Eighty-one percent of the sampled structures (n = 149) had been severely damaged or destroyed due to processes of general and local scour, sediment deposition, and channel shifting. Of the structures that were still intact (n = 43), only 31% provided the desired deepwater refuge fish habitat. These results indicated that many instream habitat structures built in southwestern Alberta were substantially degraded by small flood events, and most did not survive a sizeable flood. Our experience indicates that instream habitat structures can have a short-term benefit in providing habitat during the period immediately following a localized stream disturbance. Structures must be appropriately located and designed if they are to withstand even minor flood discharges. Regular maintenance will be required for them to remain effective. Our research suggests that instream habitat structures tend to be ephemeral and they do not necessarily provide useful habitat over the long-term.

Pattenden, R., Miles, M., Fitch, L., Hartman, G., and Kellerhals, R. 1998. Can instream structures effectively restore fisheries habitat? Pages 1-11 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

Instream structures have been used to restore and improve fish habitat in streams of southwestern Alberta since the 1970s (R.L. & L. Environmental Services Ltd. 1993). Many of these projects were designed to create deepwater refuge, a habitat type considered to be a limiting factor to fish during low-flow periods. The projects were undertaken by government, industry, and volunteer organizations. They were used to: mitigate the effects of riparian or instream development; compensate for habitat lost during instream construction; restore the effects of historical land use activities; and enhance habitat for sportfish. Native sportfish occurring in these streams include cutthroat trout (*Oncorhynchus clarki*), bull trout (*Salvelinus confluentus*), and mountain whitefish (*Prosopium williamsoni*). Naturalized populations of rainbow trout (*Oncorhynchus mykiss*), brook trout (*Salvelinus fontinalis*), and brown trout (*Salmo trutta*) also reside in many of the drainages.

Between 1991 and 1992, an audit of several hundred habitat structures was undertaken to assess the structural stability, habitat suitability, and maintenance requirements of structures in southwestern Alberta streams [Study One] (R.L. & L. Environmental Services Ltd. 1993). During the winter of 1993–1994 the audit information was supplemented with air photo and other office analyses [Study Two] (Fitch et al. 1994). This work attempted to identify fluvial and hydraulic characteristics that influenced structure performance. In June 1995, streams in southwestern Alberta were subjected to a sizeable flood (>100 year return period). Following this event, a representative sample of the previously inventoried structures were re-evaluated to assess their post-flood performance [Study Three] (R.L. & L. Environmental Services Ltd. et al. 1996). At the same time, factors that affected structure performance were identified.

This paper presents a summary of the results of these investigations, and discusses the relevance of these and other findings to the use of instream structures as a habitat enhancement and restoration technique in southwestern Alberta streams.

Study Area

The study area is located in southwestern Alberta (Fig. 1). The studied streams are all located within the Oldman River drainage, which encompasses three physiographic regions: the Rocky Mountains, Rocky Mountain Foothills, and Alberta Plains (Bostock 1967). Habitat structures are present on 26 streams in the study area, and these streams

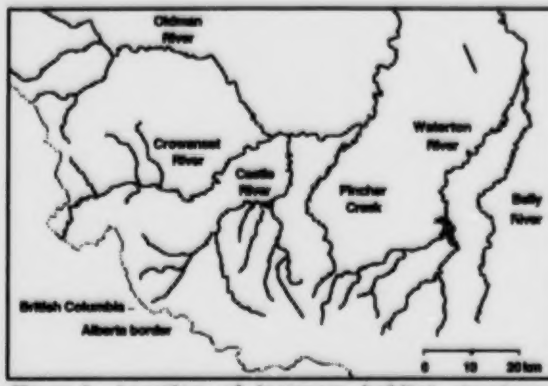

Figure 1. Location of instream habitat structures within the Oldman River sub-basin, southwestern Alberta.

exhibit a range of channel characteristics (R.L. & L. Environmental Services Ltd. et al. 1994). These include variable stream size, gradient, channel stability, and bedload transport rates. Streamflows are dominated by the spring snowmelt freshet (Fisheries and Environment Canada 1978). Discharges typically peak in May and June, and then decrease in summer and winter. Rain-on-snow events occur infrequently in the spring and produce the floods of record (Nemanishen 1979). Such an event occurred on June 5 and 6, 1995 (S. Lowe, Alberta Environmental Protection, River Engineering Branch, 8215 – 112 Street, Edmonton, Alberta T6G 5A9, personal communication). Warm temperatures initiated rapid snowmelt, which was immediately followed by intense rainfall. This resulted in significant flood discharges that were well above typical annual peak flows (Table 1). The flood was thought to have a >100 year return period (S. Lowe; personal communication).

Methods

Study One

Field evaluations of the structures undertaken between 1990 and 1991 followed procedures described in R.L. & L. Environmental Services Ltd. (1993). Each structure was rated by an experienced observer using a subjective scale [1 (best) to 4 (worst)]. The observers rated structural stability based on the structure design and function, whereas habitat suitability was rated based on the quality of the habitat provided (Table 2). The biological goal of the structures was to enhance the quality and quantity of deepwater refuge habitat suitable for adult trout during low-water periods.

Table 1. Relative magnitudes of June 1995 flood event in selected streams in southwestern Alberta compared with previously observed maximum instantaneous discharges values

Stream	Years of record	Pre-June 1995 maximum observed instantaneous peak discharge (m^3/s)	June 1995 peak discharge (m^3/s)	Estimated return period (year)
Pincher Creek	34	172	250	200–500
Crowsnest River	41	74	135	> 500
Castle River	46	736	1450	> 500
Racehorse Creek	28	89	245	> 1000

Table 2. Performance rating criteria used during evaluations of instream habitat structures in southwestern Alberta (R.L. & L. Environmental Services Ltd. 1993)

Category	Class	Criteria
Structural stability	One	Structure retains its original design and function
	Two	Structure has minor faults in its original design and function
	Three	Structure has major faults in its original design and function
	Four	Structure failed to retain its original design and function
Habitat suitability	One	Highly effective in meeting target habitat
	Two	Moderately effective in meeting target habitat
	Three	Low effectiveness in meeting target habitat
	Four	Not effective in meeting target habitat

Study Two

Twenty hydraulic and 15 fluvial variables were compared with structure performance in 1993 and 1994. Detailed descriptions of these variables and the methods used in their analyses are provided in R.L. & L. Environmental Services Ltd. et al. (1994) and Fitch et al. (1994). Hydraulic variables were generated using long-term flow records for study area streams. Fluvial variables were derived from the interpretation of air and ground photos. Statistical analyses were undertaken to assess whether there was a relationship between the hydraulic and fluvial variables and the structure's physical stability and the suitability of the resulting habitat. Prior to analyses, data were grouped by structure type (boulders, groynes, revetments, and weirs).

Study Three

Field evaluations of the post-flood condition of structures in 1995 followed procedures described in R.L. & L. Environmental Services Ltd. et al. (1996). Structural stability was rated at each site by an experienced observer using a subjective scale [1 (best) to 4 (worst)]. A subsample of these structures was also rated for habitat suitability to fish using the same approach. Detailed inspections were completed to determine how variations in channel morphology, hydraulic regime, and structure design affected post-flood structure performance.

Results

Study One

A total of 351 habitat structures were inventoried. Ninety percent of these structures were built between 1985 and 1990. The types of structures were diverse, but the sample consisted primarily of four basic types: weirs (38%), boulders (23%), revetments (14%), and groynes (10%). Of these, approximately half were present in combination with other structures. To avoid biases resulting from interactions amongst multiple structures, performance analyses were restricted to solitary structures associated with the four basic types ($n = 174$).

Structural stability performance ratings varied within each of the structure types. Ratings for boulders, groynes, and revetments were significantly skewed towards higher performance values (Table 3). In contrast, the distribution of ratings for weirs did not differ significantly from random. When data for all structure types were grouped, 63% of the sample received a rating of Class Two or better.

Structural performance was dependent on structure type (Fig. 2). Boulder, groyne, and revetment structures achieved a mean structural stability rating of 1.9 or better, but the mean rating of weir structures was much lower (2.6). This difference was statistically significant for boulders and weirs, and for groynes and weirs.

In terms of habitat suitability within each structure type, rating distributions for revetments and weirs were significantly skewed towards higher values (Table 4). However, the rating distributions for boulders and groynes did not differ from random. As with findings for structural stability, when habitat suitability ratings data for all structure types were grouped, the majority were rated as Class Two or better (61% of sample).

Habitat suitability mean ratings for structure types ranged between 2.2 (boulders) and 2.4 (revetments) (Fig. 2). Statistical comparisons of mean habitat suitability ratings indicated no significant differences between structure types.

These results suggest that during the short-term (between 2 and 7 years), approximately 40% of surveyed structures did not perform satisfactorily. This high level of failure occurred despite the majority of structures having only been exposed to floods with <6 year return period.

Study Two

Statistical analyses of structure performance relative to hydraulic and fluvial variables were conducted using solitary structures belonging to one of three categories: boulders ($n = 45$), groynes ($n = 20$), and weirs ($n = 93$).

There were no statistically significant differences recorded (t-test, $p>0.05$) for any of the investigated hydraulic variables in each of the structure categories (Table 5). (Note: Statistical comparisons made using combined data: structures with ratings of one and two compared to structures with ratings of three and four). The results do not necessarily imply that the investigated hydraulic parameters were unimportant. The data may simply be insufficient to identify statistically significant relationships, or the causes of failure may be too confounded to be identified by the analytical procedures used.

However, analyses did suggest some factors deserve further investigation. First, groyne

Table 3. Frequency of structural stability ratings and overall percent distribution for instream habitat structures inventoried in southwestern Alberta between 1991 and 1992

Category	Class				Statistical
	One	Two	Three	Four	significance
Boulders	31	7	3	4	<0.001
Groynes	10	5	4	1	<0.05
Revetments	7	3	6	0	<0.05
Weirs	25	22	30	16	ns
Overall percentage	42	21.3	24.6	12.1	100

Note: Statistical significance of rating distribution based on G-test for goodness-of-fit ($p<0.05$).

Table 4. Frequency of habitat suitability ratings and overall percent distribution for habitat structures inventoried in southwestern Alberta between 1991 and 1992

Category	Class				Statistical
	One	Two	Three	Four	significance
Boulders	16	14	7	8	ns
Groynes	3	9	6	2	ns
Revetments	1	9	4	2	<0.05
Weirs	25	30	26	12	<0.05
Overall percentage	25.9	35.5	24.6	14	100

Note: Statistical significance of rating distribution based on G-test for goodness-of-fit ($p<0.05$).

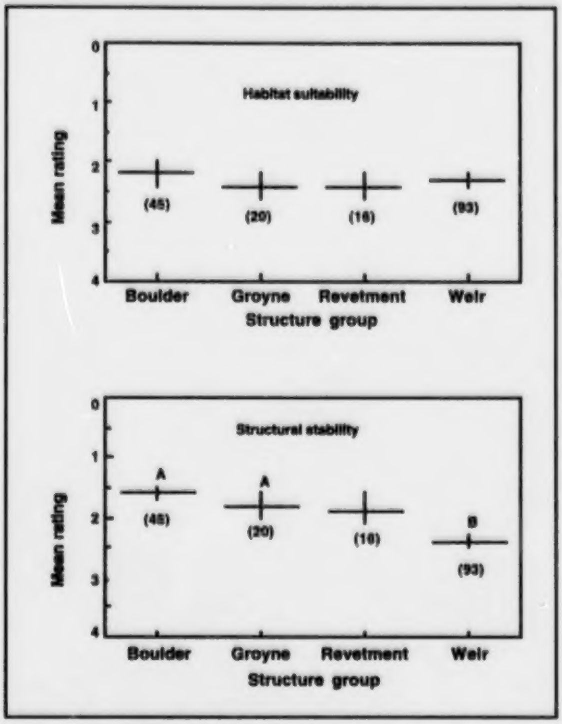

Figure 2. Mean structural stability and habitat suitability ratings ±1SE (*n*) of instream habitat structures inventoried in southwestern Alberta between 1991 and 1992. Statistical significance based on Duncan Means Test; different letters designate significant differences between means ($p<0.05$).

performance was highest at sites where the ratio of the 90th percentile bed material size to average bed size (d90/d50) exceeded 2.2. Also, boulders exhibiting a B-axis width to average bed material size (B-axis/d50) ratio of ≥25 had a higher likelihood of remaining stable. Finally, data for weirs indicated that the ratio of maximum water depth above the weir divided by the weir height (HWpp/P) was positively related to weir structural stability. This result suggests that weirs placed in locations with high stable banks are less likely to be outflanked.

The results of the fluvial analyses indicated that the physical setting significantly affected both the structural stability and habitat suitability of boulders and weirs (Table 6). Boulder structures were affected by nine fluvial variables, and weir structures by 11 variables. Of the 15 variables examined, three produced consistent results: the rate of bedload transport (negative relationship), channel stability (positive relationship), and sinuosity (negative relationship). No statistically significant relationships were found for groynes. The lack of any statistical relationships for groyne structures may indicate that variation in design and construction technique masked any local effects arising from differences in channel characteristics.

From these results, it appears that boulder and weir structures perform better in vertically and laterally stable stream channels that transport small amounts of coarse-textured sediment. A bank height >2 m is also desirable, as it increases the lateral stability of the channel. Small bed material size beneficially affects habitat suitability at boulder sites, possibly because the stream can periodically mobilize the channel bed. In some circumstances, small bed material size may also be an indicator of low stream power (Dingman 1984), which increases the likelihood of boulders remaining in place during flood events.

Study Three

The post-flood assessment of structure stability was undertaken at 149 structures. The results indicate that 131 (87%) of the structures were severely damaged or destroyed during the June 1995 flood (Table 7). The results of habitat suitability ratings, which were based on a sample size of 43 structures, were similar. Twenty-nine (67%) of the sampled structures provided little or no useful deepwater refuge habitat for adult fish.

A comparison of pre- and post-flood results for inventoried structures showed a strong downward trend in performance following the June 1995 event (Fig. 3). While the distribution of pre-flood ratings found in Study One were skewed towards increased performance, the opposite distribution was recorded following the flood. This pattern was apparent for both structural stability and habitat suitability.

Discussion

Comparison to Other Studies

The Oldman River Dam Mitigation Project in southwestern Alberta has undertaken an ambitious program to construct habitat enhancement structures in the same geographic area as the present study. As of May 1990, completed projects included 204 structures on the Crowsnest River, 18 structures on the Oldman River, and 12 structures on the Castle River (Pisces Environmental Consulting Services Ltd. 1991). The design criterion for the

Table 5. Description of hydraulic variables used to evaluate the performance of instream habitat structures in southwestern Alberta

Variable	Description
H_{pp}/B	Ratio of peak mean flow depth and boulder diameter
$V_{pp}^3(g^3H_{pp}d_{50}^2)^{0.5}$	Measure of the ratio of forces acting to move bed material over the forces acting to stabilize bed material
H_{pp}/d_{50}	Measure of bed material stability using peak mean flow depth
$V_{pp}/(g\,H_{pp})^{0.5}$	Flow Froude number
B/d_{50}	Ratio of boulder diameter to mean bed material size
L_s/B	Measure of the relative spacing of boulders in groups
% W	Placement of a boulder within a cross-section
V_{pp}^2/gB	Measure of the ratio of forces acting to move a boulder over the forces acting to stabilize the struc ture
V_{pp}^2/gd_{50}	Measure of the ratio of forces acting to move bed material over the forces acting to stabilize bed material
W_c/W	Relative constriction caused by weirs
L_s/L_c	Ratio of two measures of groynes which provides indication of interaction between groyne structures
L_p/W	Ratio of bank protection length to stream width
HW_{pp}/P	Relative flow depth over weirs
H_{cs}/H_w	Measure of flow amount concentrated over the lowest point of a weir
B_w/B	Defines the width of a weir crest in terms of numbers of boulder diameters
d_{90}/d_{50}	Measure of the gradation of bed material
Angle to flow	Angle between the direction of flow and the centerline of groyne, which increases as the groyne points farther downstream
V^2/gB	Measure of the ratio of water force to gravitational force on a boulder
Bank angle	Angle from the upstream bank to the arm of a V-weir
Nose angle	Interior angle of the V in a V-weir

Table 6. Summary of the relationships between fluvial characteristics and structural stability and habitat suitability ratings of boulder and weir structures in southwestern Alberta

Characteristic	Category			
	Boulder		Weir	
	Structural stability	Habitat suitability	Structural stability	Habitat suitability
Bed material size	ns	-	ns	ns
Channel confinement	+	+	+	+
Bedrock exposure	ns	+	ns	ns
Bank height	-	ns	-	-
Rate of bedload transport	-	-	-	-
Vertical instability	-	-	±	±
Lateral instability—style	-	-	ns	-
Debris frequency	ns	-	ns	ns
Sinuosity	-	-	-	-
Bank material stability	+	+	+	+
Bar forms	ns	ns	ns	-
Lateral instability—rate	ns	ns	ns	-
Lateral channel pattern	ns	ns	ns	-
Valley flat width	ns	ns	-	-
Number of significant factors	7	9	7	11

Note: + = implies increased performance; – = implies decreased performance; ns = not significant. Statistical significance based on Kruskal-Wallis test ($p<0.1$) for most parameters. Statistical significance for bed material size based on Tukey's test ($p<0.05$).

Table 7. Summary of structural stability and habitat suitability ratings for instream habitat structures inventoried in southwestern Alberta streams following the June 1995 flood

Rating	Structural stability		Habitat suitability	
	Number	Percentage	Number	Percentage
Class 1	18	12.1	6	13.9
Class 2	10	6.7	4	9.3
Class 3	14	9.4	7	16.3
Class 4	107	71.8	26	60.5
Total	149	100	43	100

Oldman Mitigation Program structures was 100 year return period. Design drawings are shown in Lowe (1992).

In October 1992, the overall physical stability success rating was 84% for these structures. However, following the June 1995 flood, the physical stability success rating fell to 49% (J. Englert, Alberta Public Works Supply and Services, Civil Projects Branch, 8215 – 112 Street, Edmonton, Alberta T6G 5A9, personal communication). This value is considerably higher than the 19% physical success rate found in our study. It likely illustrates the importance of engineering advice during the design and construction of enhancement projects. The higher value may also relate to a different method of rating the physical stability of structures.

A recent study in British Columbia evaluated the success of 99 habitat improvement projects (Hartman and Miles 1995). Only 55% of the projects were found to be physically successful. Biological success was estimated as 50%, despite a general lack of any detailed post-construction biological evaluation. The average post-construction assessment period was only 2 years. The longer term success statistics will likely be lower, due to physical deterioration of the structures.

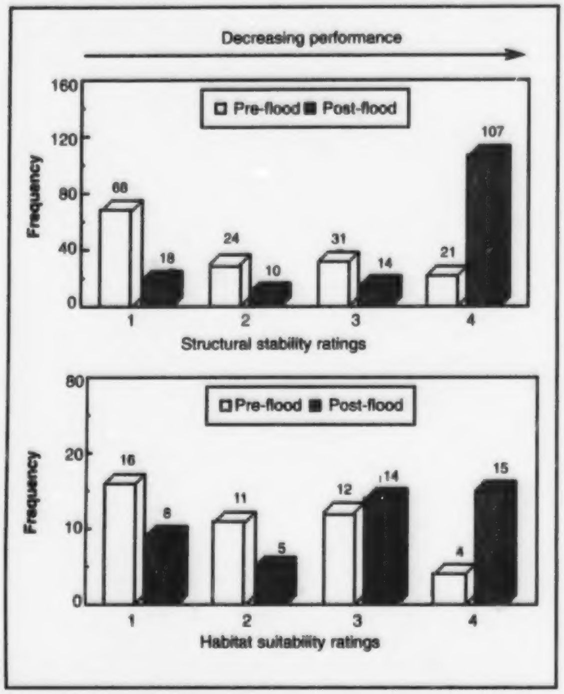

Figure 3. Comparison of pre- and post-flood performance ratings of structural stability (n = 149) and habitat suitability (n = 43) for structures inventoried in southwestern Alberta.

A review of other habitat enhancement projects indicates that the British Columbia experience is not unusual (Miles, in press). In Alaska, habitat restoration projects have a 47% success rate (Parry et al. 1993; Parry and Seaman 1994). A review of projects in Oregon and Washington found a 40% success rate (Frissell and Nawa 1992).

A large number of instream habitat improvement projects have been undertaken in the Willamette, Mt. Hood, and Snoqualmie national forests in the Pacific Northwest (Oregon and Washington). Studies by Uthank (1994), Higgins and Forsgren (1987), and Doyle (1991) have reported over 80% success rates. These high success rates appear to result from the following: i) use of log structures that were designed "to function with the natural tendencies of the stream flow and hydrologic functions" (Uthank 1994); ii) design improvements resulting from extensive experience in a geographically limited area; iii) an annual inspection and maintenance program; and iv) success criteria that appear to be mainly based on structure presence or absence.

Biological studies, such as those by Hunt (1976), Ward and Slaney (1981), and Ward (1993) indicate that, in some circumstances, instream structural enhancement can increase fish production. However, there is increasing evidence that structural measures alone do not necessarily improve fish production. For example, ongoing studies for the Oldman River Dam Project in Alberta have not demonstrated that habitat enhancement structures have significantly increased trout production (O'Neil and Pattenden 1994). Beschta et al. (in press) reviewed the biological success of several stream restoration projects in the northwestern United States. In the majority of cases, the production of anadromous fish was not improved. Riley and Fausch (1995) documented an increase in fish numbers and biomass in enhanced sections of six northern Colorado streams. However, the authors suggested that this success was related more to the movement of fish into structures from adjacent areas, rather than an increase in fish production (i.e., growth or survival).

Implications for Future Projects

From a physical perspective, the results from other studies show that structural measures that attempt to physically create stream habitat frequently do not perform well. The analyses undertaken for this project indicate that success rates are likely to be poorer in higher energy environments, or on streams that are either laterally unstable or are carrying high sediment loads (Fitch et al. 1994). Unfortunately, most land-use changes accelerate channel instability and elevate rates of sediment transport, which decreases the likelihood of successful structure performance.

These results do not indicate that structural measures to enhance or restore fisheries habitat should not be undertaken. However, they do indicate that enhancement structures are frequently ephemeral, and are therefore best suited as temporary measures to enhance fisheries productivity over the short-term. This result should not be unexpected, as habitat features in a non-impacted alluvial stream are constantly being formed and destroyed due to the processes of river erosion and channel shifting. The longer-term solution to restoring or creating fisheries habitat is to re-establish or maintain the fluvial processes that are responsible for habitat formation (Kellerhals and Miles 1996). At a minimum, this requires that river corridors be established that allow stream channels to freely migrate. Riparian vegetation needs to be protected, and in areas where it has been removed, efforts should be made to assist its

recovery. Depending on the watershed, it may also be necessary to take measures to reduce sediment production, and to expedite the hydrologic recovery of clear-cut or other areas that could be increasing the size of flood discharges (Booth 1991; Jones and Grant 1996).

This approach to habitat creation by "re-naturalization" of a river implies that it may be difficult or impossible to enhance fisheries production in pristine, non-impacted streams using physical structures. Managers must recognize that there are physical limits to the amount of a particular habitat a river is capable of maintaining. For example, fisheries biologists view deep water or pool habitats as limiting factors that should be increased in number. In an alluvial river, pools occur with a size and frequency that is dependent on the meander wavelength, which in turn is a property of the hydraulic regime (Bray 1982). These relationships cannot be changed. It is likely that organisms that make up a stream's ecosystem are adapted to these comparative frequencies. Attempts to artificially manipulate this relationship, for perceived "icthyocentric" purposes, have a high probability of failure because of the complex inter-relationships between the physical environments and the biological organisms that inhabit them. Questions like "How much riffle area is needed to support the organisms that provide food for fish that live in pools?" are difficult to answer. Many projects, therefore, appear to be undertaken without a solid understanding of the biological limiting factors, or a sound basis for predicting the results of proposed habitat manipulations. These uncertainties are at least partially responsible for the low success rates recorded during this study.

Conclusions

Humankind is changing river morphology and associated aquatic habitats through direct (or local) interference, and by altering the formative external factors, such as stream flow, sediment loading, and bank characteristics (Kellerhals and Miles 1996).

Our experience indicates that instream structures can have short-term habitat benefits during the period immediately following a localized stream disturbance. Structures must be appropriately located and designed if they are to withstand even minor flood discharges. Regular maintenance is required if they are to remain effective. However, instream habitat structures should not be viewed as a long-term solution for the development of good fish habitat.

Evidence continues to accumulate indicating that fish habitat improvement projects are often physically unsuccessful. The failure rate is particularly high for projects that rely on some form of habitat construction or artificially maintained channel morphology. Performance also tends to be comparatively poor in high energy environments, in laterally or vertically unstable channels, or on gravel-bedded streams that transport significant quantities of bedload. Better design practices might improve performance, but experience indicates it is very difficult to restore a river once it has been severely altered.

A variety of channel processes are required to maintain a healthy stream.Over the long-term, stream health and habitat availability can only be achieved by preserving or re-establishing these processes. This might involve watershed-level restoration activities, limiting the amount of development in sensitive watersheds, and establishing appropriately sized riparian corridors. These kinds of activities require community cooperation, integrated planning, and, possibly, the compensation of landowners for required changes in land use. This is much more complicated than merely placing a few logs or rocks in a stream channel. The challenge is to find ways to undertake this work in the extensive areas that require restoration.

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Human Impact on Aquatic and Riparian Ecosystems in Two Streams of the Thompson River Drainage, British Columbia

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Abstract

This study is a short-term ecosystem assessment of spatial changes in aquatic and riparian habitat, invertebrates, and vertebrates in relation to the intactness of riparian vegetation along two agriculturally impacted streams of inland British Columbia. Habitat assessment characterized riparian-floral, channel-complexity (lateral), and hydraulic conditions. The biota assessed included riparian vertebrates (especially frogs and birds), riparian invertebrates (shoreline and emergent taxa), drift and benthic invertebrates, aquatic megainvertebrates, and fishes. Biological indicators of deforestation were developed via habitat diversity, biotic integrity, biodiversity, and guild analyses. The results showed that semi-forested sites were consistently higher in habitat and biological diversity than forested and/or unforested (shrubby or grassy) sites, for both aquatic and riparian taxa. In contrast, the highest densities of fish foods (drift and benthic invertebrates) were found at unforested sites, whereas aquatic megainvertebrates were most abundant at semi-wooded (semi-forested or shrubby) sites and fish densities were often highest at treed (forested or semi-forested) sites. Smaller invertebrates and riparian vertebrates were classified into treed, generalized, or unforested guilds based on their abundance across sites, whereas fish and aquatic megainvertebrates were classified into wooded (non-grassy) or unforested guilds. Fish and megainvertebrate taxa were also classified into one of four trophic guilds, the two anadromous guilds (drift insectivores and drift herbivores) being more common at sites with woody riparian zones and the two resident guilds (benthic insectivores and benthic omnivores) containing taxa common at sites with woody or unforested riparian zones. The results suggest that semi-forested riparian conditions are conducive to biodiversity maintenance in stream valleys of British Columbia's dry interior, perhaps because fire and other natural disturbances have historically provided patchy forest conditions in floodplain habitats.

Vadas, R.L., Jr. 1998. Human impact on aquatic and riparian ecosystems in two streams of the Thompson River drainage, British Columbia. Pages 13–30 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

The importance of riparian-buffer strips along streams to protect habitat quality for riparian and aquatic faunas, including salmonid fishes and their invertebrate foods, is well recognized in the U.S. Pacific Northwest (PNW) (Krygier and Hall 1971; Sedell and Swanson 1984; Salo and Cundy 1987; Raedeke 1988; Shaw and Bible 1996), elsewhere in the U.S. (Brinson et al. 1981; Ohmart and Anderson 1982; Likens 1985), and in western Canada by Trout Unlimited (Brewin 1992, Brewin and Monita 1998; Hamilton 1992) and other researchers (Morgan and Lashmar 1993; Pearce 1993; SALASAN Associates and Dovetail Consulting 1995; Nener et al. 1997). Indeed, riparian-floral cover has often been incorporated into suitability models to describe the habitat needs of fish and wildlife (Terrell et al. 1982; Graves and Dittberner 1986). Although buffer strips under 30 m wide often protect water quality and aquatic invertebrates (Clinnick 1985; Ontario Ministry of Natural Resources 1988; MacDonald et al. 1991; Vadas 1998b), such vegetation is not adequate for preserving aquatic and riparian ecosystems (Burke and Gibbons 1995; Waters 1995; Vadas and Newman 1998).

Ecosystem studies to document the effect of deforestation and other riparian impacts are now being done to assess changes in aquatic and riparian physiochemical conditions, vegetation, invertebrates, and/or vertebrates in British Columbia (B.C.) (Macdonald and Scrivener 1992; Bernard et al. 1994; Tschaplinki 1996) and elsewhere in North America (Maki et al. 1975; Szaro and Rinne 1988; Gregory et al. 1991; Scruton et al. 1995). Such assessments include examination of whole fish, aquatic-invertebrate (Angermeier and Karr 1994; Cash 1995; Culp et al. 1997) or wildlife assemblages (Schroeder 1987, 1989; Schroeder and Allen 1992), including biotic integrity and biodiversity assessments, rather than the traditional focus on individual species. These analyses are often simplified by classifying animal species into trophic, habitat-use, and other ecological guilds, i.e., groups of species using similar resources along a specified niche axis such as food or cover (Hawkins and McMahon 1989; Simberloff and Dayan 1991).

In B.C. (Macdonald et al. 1988; Poulin and Scrivener 1988; Keeley and Walters 1994; Mellina and Hinch 1995) and Alberta (Alke 1995; Brewin and Monita1998), deforestation impacts from logging have received more attention than other land-clearing operations such as agriculture because the focus has been on smaller (headwater) streams rather than on valley rivers. British Columbia researchers have rationalized this research bias by suggesting that smaller streams are more sensitive to deforestation (Tschaplinki 1996) and more amenable to habitat restoration (Slaney and Martin 1997), but these hypotheses have not been adequately tested. Indeed, evidence from eastern Canada suggests that agricultural impacts on aquatic invertebrates and fishes are worse than logging effects (Welch et al. 1977), such that downstream, agriculturally impacted streams deserve more study.

The present study is a short-term ecosystem assessment of spatial changes in aquatic and riparian habitat, invertebrates, and vertebrates in relation to the intactness of riparian vegetation along two agriculturally impacted streams in inland B.C. (cf. Michel 1997). In order to determine biological indicators of deforestation, habitat-diversity, biotic-integrity, and biodiversity analyses were undertaken. Animal taxa were also classified into habitat-use guilds to assess their riparian-floral needs. Trophic-guild composition across the floral gradient was examined for aquatic taxa, to assess food-web patterns.

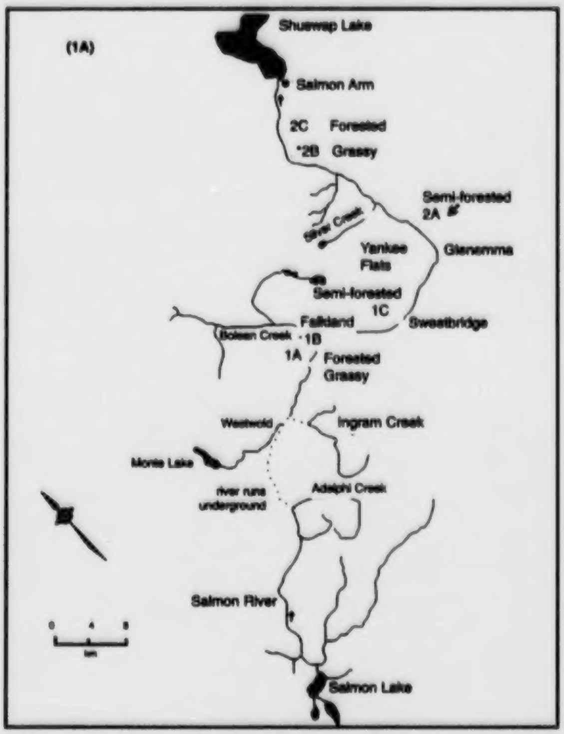

Figure 1A. Map of the Salmon River watershed, modified from Sebastian (1982). The watershed is northeast of the Nicola watershed in the B.C.-inset map shown on Figure 1B, both rivers originating near 50°10′ North and 119°45′ West coordinates. Asterisks (*) indicate reaches with obvious cattle damage.

Materials and Methods

Study Sites

The two river valleys studied, namely the mainstems of the Salmon (SR) and Nicola (NR) rivers, were in the Thompson River drainage (Fraser River basin) in the dry, southern interior of B.C. (Fig. 1A and B). These watersheds show a lateral (bank to upslope) progression from (1) willow and cottonwood to (2) sagebrush (NR only), aspen, and bunchgrass to (3) ponderosa pine to (4) mesic conifers (British Columbia Ministry of Environment and Parks 1986; Department of Fisheries and Oceans 1991a; Beeson and Doyle 1995; Miles 1995b). Logging and mining activities in headwater areas and agricultural activities and urbanization in the mainstems have disproportionately affected riparian and water quality and hydrology in these watersheds compared to nearby ones, with farming and ranching impacts being especially damaging (O'Riordan 1981; British Columbia Ministry of Environment and Parks 1986; Department of Fisheries and Oceans 1991a, b, c; Quadra Consultants Ltd. 1995; John 1996; Rood and Hamilton 1995). Heavy-metal and wood-industry effluents (NR only), nutrients, coliforms, suspended sediments, and water temperatures are elevated above natural levels in (1) mainstem SR (Bryan 1976; Gregory 1989; Department of Fisheries and Oceans 1991a; Shaw and El-Shaarawi 1995), (2) middle and lower NR (Holmes 1974, 1980; Munro et al. 1985; Grace 1987; Department of Fisheries and Oceans 1991b; Nordin and Holmes 1992; Walthers and Nener 1997), and (3) some NR tributaries (Little 1974; Holmes 1988), mainstem impacts being particularly severe downstream of the townships of Salmon Arm (lower Salmon valley) and Merritt (middle Nicola valley). Nevertheless, SR is naturally productive via apatite bedrock in its headwaters (Department of Fisheries and Oceans 1991a; Culp et al. 1997). The two nearby rivers are biologically similar (Michel 1997), in part because the upper Salmon River was formerly part of the Nicola River (Miles 1995a).

Methods

Aquatic and riparian faunas and habitats were sampled during the fall of 1994. The biota assessed included riparian vertebrates (especially frogs and birds, which were identified by sight and sound), riparian invertebrates (caught with a sweep net and aquatic-emergence traps), drift and benthic invertebrates (respectively caught with drift versus Hess nets), and aquatic megainvertebrates and fishes (caught with a seine net and DC/AC electroshocker fished above a block net). Vertebrates were identified to species. Invertebrates were identified to higher taxonomic levels (usually order, suborder, or

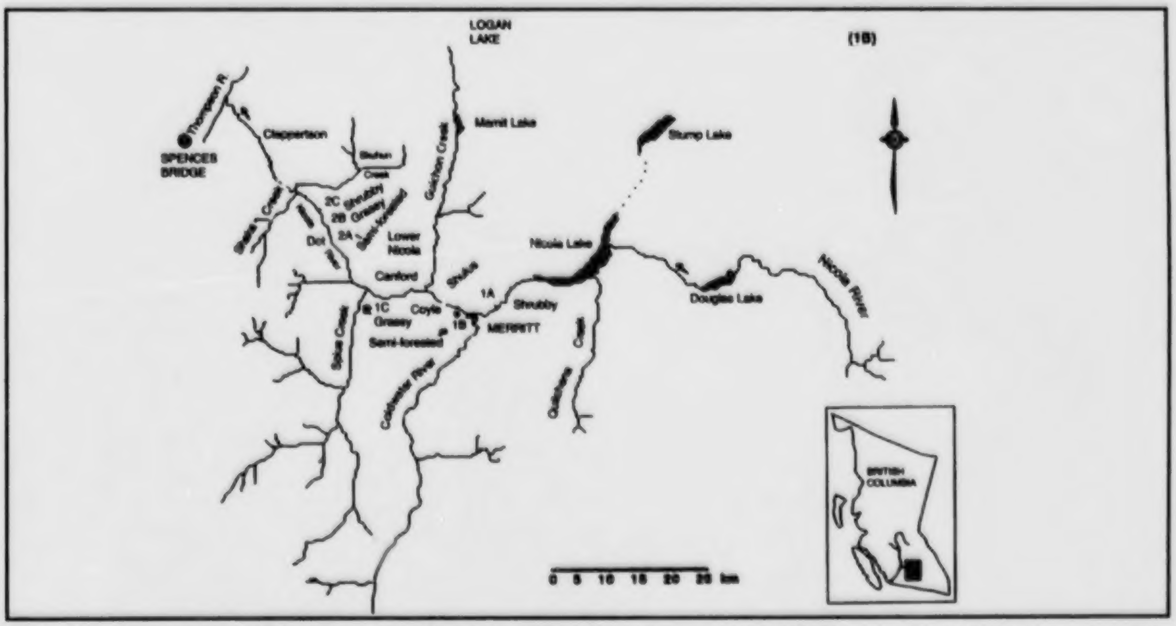

Figure 1B. Map of the Nicola River watershed, modified from Brown et al. (1979). Asterisks (*) indicate reaches with obvious cattle damage.

family). Riparian and aquatic habitats were sampled to determine the relative abundance of habitat types, which included five lower-riparian (bankside) floral-substratum categories (trees, woody shrubs, tall herbs, and bare-coarse and bare-fine substrata); five aquatic-lateral habitat types (main channels [MC], side channels [SC], backwaters [BW], and transitional habitats [MC/SC and SC/BW]); and seven aquatic-hydraulic categories (medium and shallow pools, medium and shallow runs, medium torrents, and slow and fast riffles). Further documentation of sampling methods can be found in Vadas and Orth (1993, 1998), Vadas (1998a, 1998b), and Vadas and Newman (1998). Schroeder and Allen (1992) emphasized the importance of riparian-floral, aquatic-lateral and aquatic-hydraulic diversity for lotic and riparian faunas.

For both streams, middle and lower river sites were examined to make triplet-wise comparisons among sites of differing floral intactness in the upper-riparian (floodplain) zone. The habitat types included forested (FO), semi-forested (SF), shrubby (SH), and grassy (GR); FO-SF-GR and SF-SH-GR comparisons were made in SR versus NR, respectively (Fig. 1). Based on percent-abundance and/or density (no./m^2) data and uni- and multivariate analyses (e.g., varimax-factor analysis to cluster taxa or sites), riparian and aquatic biota were classified into habitat-use guilds along the riparian-floral axis. Habitat and biological diversity were assessed with the Simpson-Levins diversity index, species richness also being calculated for vertebrate taxa. Further documentation of statistical methods can be found in Vadas (1991, 1992). Warwick and Clarke (1991) justified use of multivariate methods when examining aquatic ecosystems, because of better consistency of patterns across taxonomic assemblages than seen for univariate and graphical-distributional methods.

To determine if agricultural sites showed degraded ecological integrity, riparian-vertebrate taxa sensitive or tolerant to deforestation and aquatic-faunal taxa sensitive to pollution impacts (e.g., enrichment and sedimentation) were given special attention (Vadas 1998b; Vadas and Newman 1998). The ecological-guild classifications for riparian vertebrates were of four types. There were three crude-floral guilds, i.e., forested, semi-forested, and grassy. The seven guilds of the detailed-floral classification system were forested, treed (FO-SF), woody (FO-SF-SH), generalized, open-canopied (SF-SH-GR), unforested (SH-GR), and grassy. Species were also classified by other forest characteristics, i.e., as cavity (tree-snag) nesters (birds only) and agricultural–urban species.

The ecological-guild classifications for invertebrates were of three types. There were three macrohabitat guilds, i.e., aquatic, semi-aquatic (including emergents), and terrestrial. The five trophic categories, which were assigned herbivore:predator (H:P) ratios for food-web analyses (cf. Vadas 1990), included herbivorous (H:P = 1:0), predominately herbivorous (H:P = 2:1), omnivorous (H:P = 1:1), predominately predatory (H:P = 1:2), and predatory (H:P = 0:1). The five pollution-tolerance guilds, which were assigned points for ecological-integrity analyses, included sensitive (S = 3 points), moderate (M = 2 points), tolerant (T = 1 point), and transitional (SM = 2.5 points and MT = 1.5 points). There were four pollution indices, the first two being formulated from these points. Index I was simply the average points value across all taxa, weighted by relative abundance. Index II was the sum of points across all present taxa. Index III (percentage EPT) was the percent abundance of large-bodied, pollution-sensitive taxa, which included Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies). Index IV (EPT:D) was the abundance ratio of EPT taxa vs. dipterans (true flies).

Nekton assemblages (larger, free-moving aquatic biota) were analyzed separately for two groups, namely mesonekton (megainvertebrates and fish larvae) and larger fishes, the latter being fish efficiently caught with 6-mm mesh nets (cf. Vadas and Orth 1993). The non-trophic parameters examined included percent abundance of fish larvae (relative to megainvertebrates) and cyprinid shiners (relative to other larger fishes), the latter taxon being considered pollution-sensitive (Davis and Simon 1995). Based on examination of raw volumetric-dietary and sand-intake data, nekton taxa were tentatively classified into four trophic guilds based on whether they fed mostly above or near the bottom and upon invertebrates and/or algae; the guilds were drift insectivores, drift herbivores, benthic insectivores, and benthic omnivores (Vadas 1988, 1990).

Results

Riparian and Aquatic Habitat

Habitat diversity, which was compared in the middle and lower sections of the two rivers (four comparisons), was often highest at semi-forested sites (Table 1). Treed sites showed the highest habitat diversity in the lower-riparian zone, via greater abundance of woody vegetation. Aquatic-habitat diversity along the lateral gradient was also highest at treed sites of SR, reflecting the greater abundance of edge habitats (backwaters and/or side channels), whereas NR trends were ambiguous. Semi-forested sites showed the highest hydraulic diversity because pool, run, and riffle habitats were all abundant, whereas diversity differences among other floral-habitat types were inconsistent. The two most sedimented sites, i.e., the uppermost reaches sampled in SR and NR, were relatively low in aquatic-habitat diversity.

Riparian Vertebrates

Avian assemblages, which consisted especially of songbirds (passerines), differed among sites (Fig. 2). Higher species diversity and percent abundance of agricultural–urban species was seen at less-forested sites, whereas logging-sensitive, cavity-nesting birds (cf. Thomas et al. 1978; Steeger et al. 1995; Vadas and Newman 1998; Vadas 1998) showed highest percent abundance at semi-forested sites. Trends for species richness were inconsistent across rivers. Classification of species into floral habitat-use guilds, as defined in the literature for western North America (Vadas and Newman 1998), showed that generalists were usually dominant; treed sites did not always show greater abundance of treed guilds and lesser abundance of unforested guilds. Guild classification based on the SR-NR data set (Table 2) yielded two treed habitat-use guilds, i.e., forested (one species) and semi-forested (two species). There were also seven generalist and one grassy species. These guild classifications must be considered tentative, however, because of the limited spatiotemporal nature of the data collected for these mobile taxa.

Spotted frog adults (*Rana pretiosa*) were only found in the SR valley, where the species was omnipresent but more abundant at treed sites; densities (no./1000 m stream length) were 14.5 for semi-forested, 11.0 for forested, and 6.0 for grassy sites.

Table 1. Riparian- and aquatic-habitat diversity, based on the Simpson-Levins diversity index, in the middle (SR-M) and lower (SR-L) Salmon River and the middle (NR-M) and lower (NR-L) Nicola River. Upper-riparian (floodplain) conditions included forested (FO), semi-forested (SF), shrubby (SH), or grassy (GR).

Site	Order of habitat diversity		
	Riparian	Aquatic-lateral	Aquatic-hydraulic
SR-M	SF > FO > GR	FO = SF > GR	SF > FO > GR
SR-L	FO > SF > GR	SF > FO > GR	SF > GR > FO
NR-M	SF > SH > GR	GR > SF > SH	GR > SH > SF
NR-L	SF > SH > GR	SH > GR > SF	SF = SH > GR

Table 2. Habitat-use guilds for riparian birds along the riparian-floral axis, based on multivariate (varimax-factor) analysis of percent-abundance data. See Table 1 for abbreviations.

I. **Treed**

A. FO = yellow-rumped warbler (*Dendroica coronata*)

B. SF = black-capped chickadee (*Parus atricapillus*) and cedar waxwing (*Bombycilla cedrorum*)

II. **Generalized** = belted kingfisher (*Ceryle alcyon*), American robin (*Turdus migratorius*), American crow (*Corvus brachyrhynchos*), black-billed magpie (*Pica pica*), Brewer's blackbird (*Euphagus cyanocephalus*), barn swallow (*Hirundo rustica*), and common merganser (*Mergus merganser*)

III. **Grassy** = common yellowthroat (*Geothlypis trichas*)

Table 3. Comparisons of invertebrate taxonomic diversity across riparian-floral sites. See Table 1 for format and abbreviations.

Site	Order of taxonomic diversity		
	Riparian	Drift	Benthic
SR-M	FO > SF > GR	SF > FO > GR	FO > SF = GR
SR-L	GR > FO > SF	SF > FO > GR	FO > GR > SF
NR-M	SF > GR > SH	GR > SF > SH	GR > SF > SH
NR-L	SF > SH > GR	SF > SH > GR	SH > SF = GR

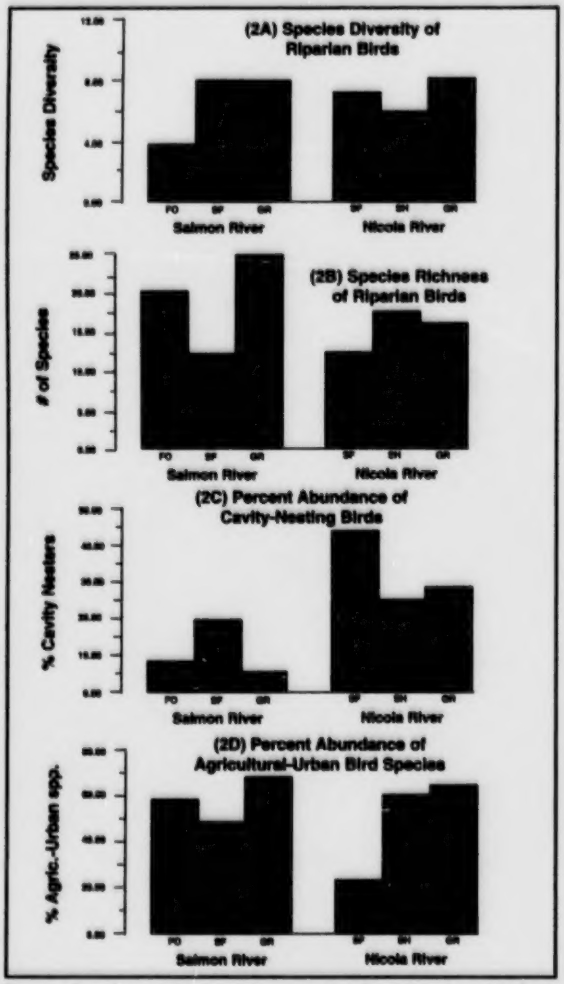

Figure 2. Assemblage parameters for riparian birds in the Salmon and Nicola River valleys, including (A) species diversity (Simpson-Levins index), (B) species richness, and numerical percentages of (C) cavity-nesting (logging-sensitive), and (D) agricultural–urban species. Increasing deforestation for a given river is from the left to right bars. FO = forested, SF = semiforested, and GR = grassy.

These data tentatively suggest that the species is in the treed guild. The abundance of spotted frogs contrasts with the species' decline farther south in B.C.'s southern-interior (Okanagan region) and western areas of Washington and Oregon, where pesticide contamination (J.E. Elliott, Canadian Wildlife Service, personal communication), water management, riparian and wetland damage, and/or bullfrog introductions have been detrimental (Phillips 1990; Orchard, 1992, personal communication; McAllister et al. 1993).

Micro- and Macroinvertebrates

Invertebrate assemblages, which were classified by ecological characteristics based on literature data (Vadas 1998), also differed among sites; taxonomic diversity was generally higher for more-forested conditions (Table 3). Riparian invertebrates were most diverse at treed sites in three of four comparisons; the general order was FO > SF > SH = GR. Drift invertebrates were also most diverse at treed sites in three of four comparisons; the general order was SF > FO > SH = GR. Benthic invertebrates showed the highest diversity at forested sites in SR, but differences among less-forested sites were inconsistent. The percent abundance of terrestrial invertebrates was inconsistent across samples (Fig. 3); whereas drift samples showed highest values at grassy sites, riparian samples showed lowest values for grassy (SR) or shrubby sites (NR).

In contrast, invertebrate density in the river was generally higher for agricultural (grassy) conditions (Table 4). Drift invertebrates were more dense at grassy sites in three of four comparisons; the general order was GR > FO > SF = SH. Benthic invertebrates were also more dense at grassy sites in SR, but differences among NR sites were inconsistent; the general order was also GR > FO > SF = SH.

Based on literature trophic data, herbivorous aquatic invertebrates were generally more dominant for unforested conditions (Fig. 3). Drift invertebrates

Table 4. Comparisons of total invertebrate density across riparian-floral sites for aquatic samples. See Table 1 for format and abbreviations.

Site	Order of total density	
	Drift	Benthic
SR-M	GR > FO > SF	GR > FO > SF
SR-L	GR > FO > SF	GR > FO > SF
NR-M	SH > SF > GR	SH > GR > SF
NR-L	GR > SH = SF	SF > SH > GR

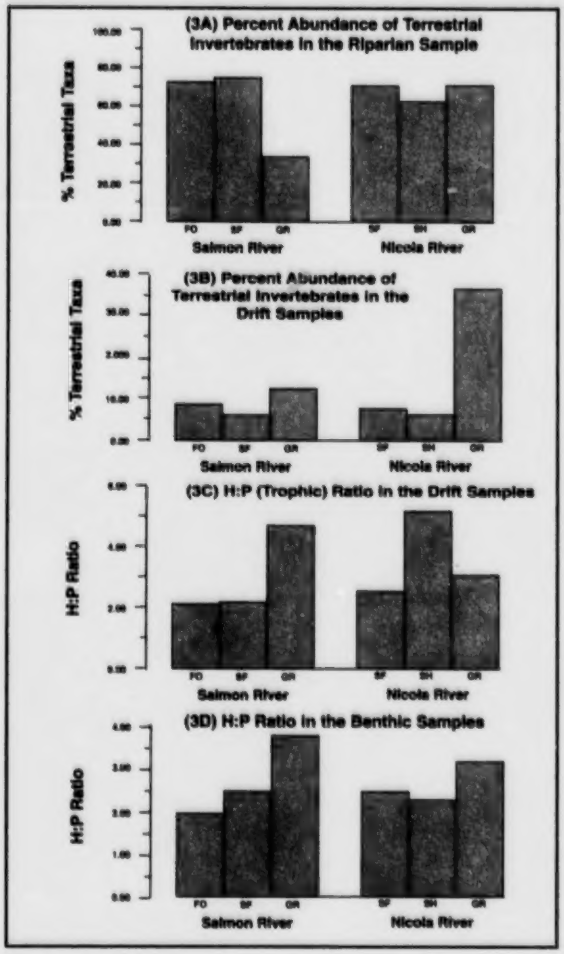

Figure 3. Relative abundance (numerical percentages) of terrestrial-invertebrate taxa (A–B) and invertebrate herbivore-predator (H:P) ratios (C–D) in the riparian and drift samples. H:P ratios were only calculated for aquatic taxa, based on the trophic classifications of Vadas (1990). Increasing deforestation for a given river is from the left to right bars. FO = forested, SF = semi-forested, GR = grassy, and SH = shrubby.

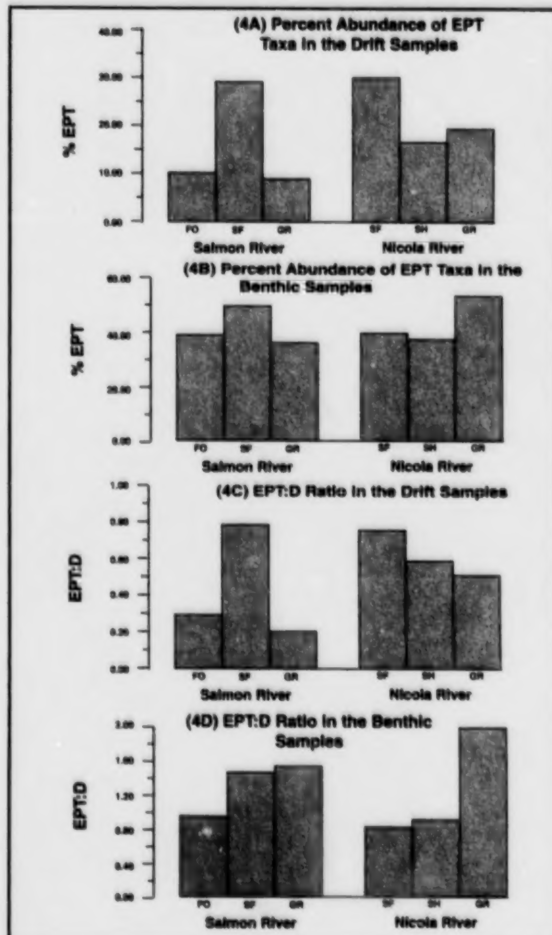

Figure 4. Percent abundance (A–B) of large-bodied, pollution-sensitive taxa (i.e., ephemeropterans, plecopterans, and trichopterans = EPT taxa) and EPT:D (EPT:dipteran) ratios (C–D) in aquatic samples. Percent EPT was calculated across all invertebrate taxa collected. FO = forested, SF = semi-forested, GR = grassy, and SH = shrubby.

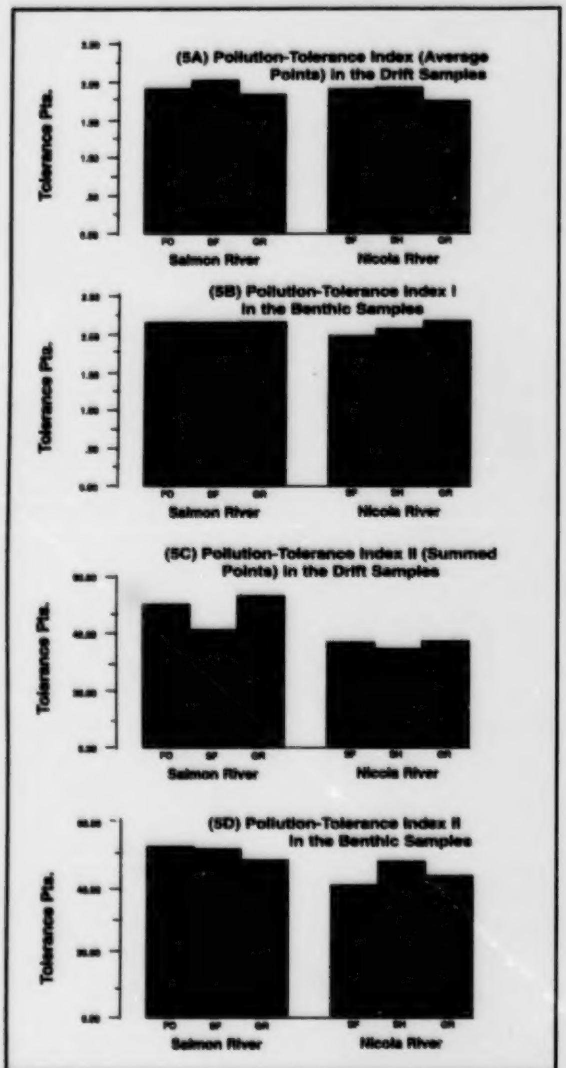

Figure 5. Pollution indices for aquatic and semi-aquatic invertebrates taxa, based on the point system defined in the text. Tolerance (A–B) averages (index I) were based on percent-abundance data (weighted point averages) and (C–D) sums (index II) were derived from presence-absence data (simple summation of points). For both indices, lower values indicate potential pollution problems. FO = forested, SF = semi-forested, GR = grassy, and SH = shrubby.

showed higher herbivore:predator (H:P) ratios at shrubby and grassy sites, whereas benthic invertebrates showed highest values at grassy sites.

Pollution indices gave ambiguous results (Figs. 4 and 5). The percent abundance of large-bodied, pollution-sensitive (EPT) taxa, was partially consistent across sites. Semi-forested sites had the highest values in SR benthic samples and SR-NR drift samples, whereas percent EPT was highest for grassy conditions in the NR benthos. The EPT:D ratio and pollution indices that require assignment of pollution-tolerance points gave inconsistent results across rivers and samples (drift versus benthos).

Floral-guild classification based on the SR-NR data set (Table 5) yielded a treed guild (three aquatic taxa), a generalist guild (eight aquatic, two semi-aquatic, and three terrestrial taxa), and an unforested guild (one semi-aquatic and four aquatic taxa). Aquatic-insect larvae and pupae and microcrustaceans were often in the treed and/or unforested guilds, suggesting that both micro- and macroinvertebrate taxa are useful as agricultural indicators. Although aquatic macroinvertebrates generally receive more research attention in ecological-integrity analyses (Root 1990; MacDonald et al. 1991; Culp et al. 1997), microinvertebrates (meiofauna) have recently been recognized as useful pollution indicators in marine and estuarine habitats (Morell 1995). Different dipteran and caddisfly taxa were in divergent floral guilds, such that even higher-level taxonomic classification would have obscured riparian trends.

Aquatic Megainvertebrates and Fishes

Nekton assemblages also differed among sites. Mesonekton diversity (megainvertebrates and fish larvae) and percent abundance of fish larvae were higher for treed sites in SR, whereas NR showed lowest values for shrubby conditions (Fig. 6). Larger fishes were most diverse at semi-forested sites in three of four comparisons, but differences among other floral-habitat types were inconsistent (Table 6). Trends for percent abundance of pollution-sensitive shiners and fish-species richness were ambiguous; redside shiners were relatively rare at semi-forested and grassy sites (Fig. 7). These results and those for riparian vertebrates (see above) suggest that species diversity is a better measure of biodiversity than species richness, largely because the latter parameter is more sensitive to sample size.

Nekton density was often lowest for agricultural (grassy) conditions. Mesonekton showed highest

Table 5. Habitat-use guilds for riparian and aquatic invertebrates along the riparian-floral axis, based on multivariate (varimax-factor) analysis of percent-abundance data (for all samples) and/or triplet-wise comparisons of density data (for drift and benthic samples). LV = larva, NM = nymph, PP = pupa, and AD = adult.

I. **Treed-aquatic** = ceratopogonid (dipteran) LV, sand-cased trichopteran (caddisfly) LV, and ostracod

II. **Generalized**
- A. **Aquatic** = ephemeropteran (mayfly) NM, plecopteran (stonefly) NM, chironomid (dipteran) LV, uncased trichopteran LV, hydracarinid (mite), oligochaete, nematode, and hydroid
- B. **Semi-aquatic** = collembolan (springtail) and nematoceran (dipteran) AD
- C. **Terrestrial** = homopteran, hymenopteran, and arachnid (spider)

III. **Unforested**
- A. **Aquatic** = lepidopteran (moth) LV, tipulid (dipteran) LV, nematoceran PP, and cladoceran
- B. **Semi-aquatic** = brachyceran (dipteran) AD

Table 6. Comparisons of taxonomic diversity and density across riparian-floral sites for larger (non-larval) fish species. See Table 1 for format and abbreviations.

	Order of parameter values	
Site	Diversity	Density
SR-M	GR > SF > FO	SF > FO > GR
SR-L	SF > FO > GR	FO > GR > SF
NR-M	SF > GR > SH	SF > SH > GR
NR-L	SF = SH > GR	SH > GR > SF

density for semi-forested sites in SR, whereas wooded sites had the highest densities in NR; grassy sites were consistently lowest (Fig. 6). Trends were inconsistent for larger fishes (Table 6); grassy sites showed lowest values in middle-river sections, whereas semi-forested sites showed lowest values in lower-river sections (where grassy sites showed intermediate densities).

Floral-guild classification based on the SR–NR data set were done on multispecific fish families and individual nekton species. The family analysis yielded two wooded guilds, i.e., generalized-forest (salmonids) and semi-wooded (minnows). Two families were generalists, namely suckers and sculpins, even though both families are typically considered pollution-sensitive (Davis and Simons 1995). The species analysis (Table 7) yielded three wooded guilds, i.e., treed-shrubby (four fish species), semi-forested (one megainvertebrate taxon), and generalized-SF (two fish species). There were four unforested guilds, i.e., generalized-shrubby (one megainvertebrate taxon), shrubby (one fish species), shrubby-grassy (two fish species), and grassy (three fish species). Because aquatic-insect nymphs and fishes were in the treed and unforested guilds, both taxa appear to be useful as agricultural indicators. As with the micro- and macroinvertebrate analysis, closely related taxa (notably, for stoneflies, salmonids, and minnows) were in divergent floral guilds.

Trophic and/or migratory patterns were evident along the floral gradient (Table 7). First, anadromous drift feeders, including both insectivores and herbivores, were most abundant at sites with woody riparian zones. Second, resident benthic feeders, including both insectivores and omnivores were most abundant at sites with woody or unforested riparian zones. Hence, plant eating, which is assumed to be more common in degraded stream habitats (see above; Davis and Simons 1995), was rare among fishes in the Salmon River and common in the Nicola River regardless of riparian intactness, because omnivorous fish species were found only in the latter river. There was also some evidence for greater feeding activity at unforested sites, as represented by higher food volumes in fish guts, in correlation with the greater food abundance at these sites (see above; Vadas 1998b).

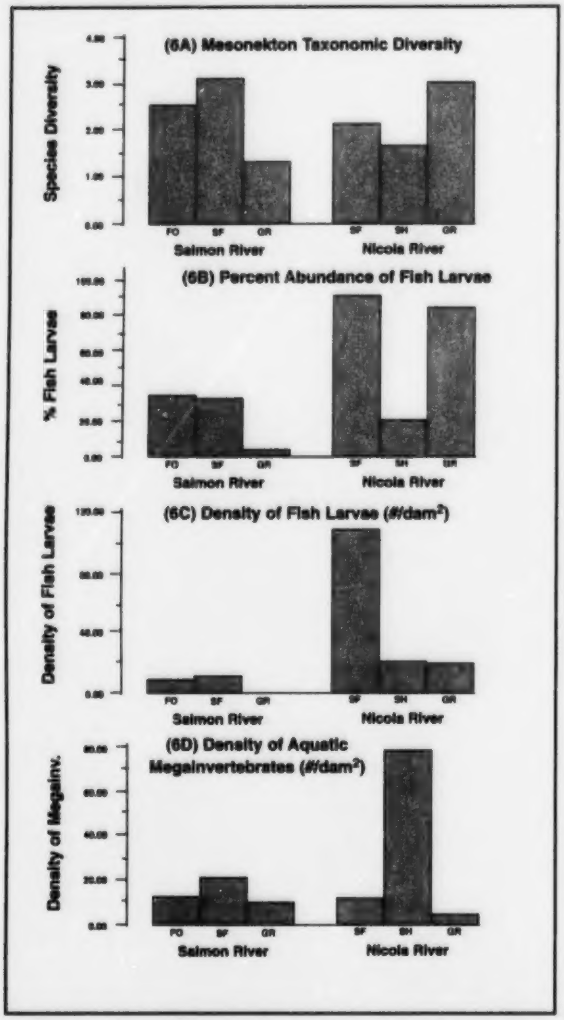

Figure 6. Assemblage parameters for mesonekton, i.e., fish larvae and aquatic megainvertebrates. The variables include (A) mesonekton taxonomic diversity, (B) numerical percent of sample composed of fish larvae (as opposed to megainvertebrates), and (C–D) density of both mesonekton groups. FO = forested, SF = semi-forested, GR = grassy, and SH = shrubby.

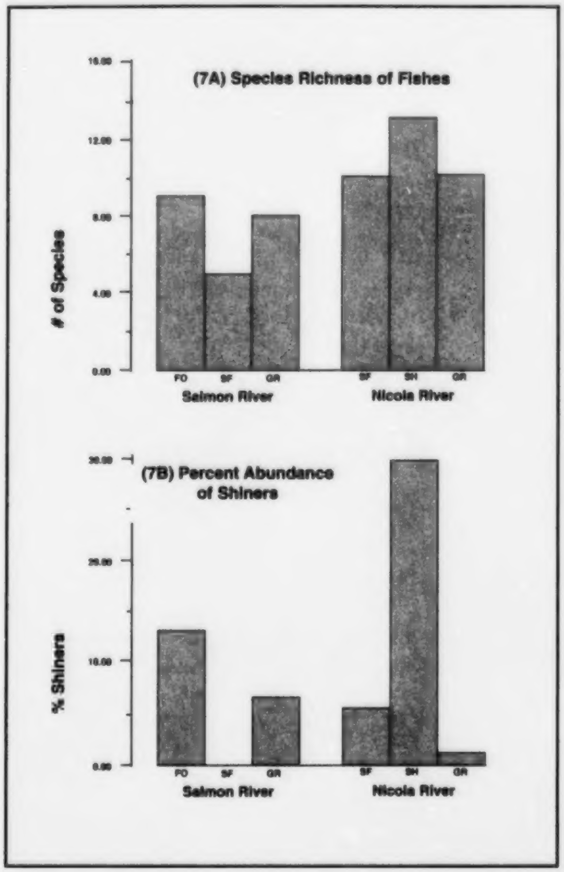

Figure 7. Assemblage parameters for mesonekton and/or non-larval (larger) fishes, including (A) fish-species richness and (B) numerical percent of large-fish samples composed of insectivorous, pollution-sensitive shiners. FO = forested, SF = semi-forested, GR = grassy, and SH = shrubby.

Discussion

To synthesize, these results suggest that semi-forested conditions are conducive to maintaining ecosystem balance; watershed biodiversity should be maintained under such conditions via increased riparian-habitat diversity (benefitting riparian fauna) and aquatic-nutrient enrichment (benefitting aquatic fauna) (Telfer 1974; Michel 1997; Vadas 1998b). It is likely that habitat and biological trends differed among taxa and were not always consistent across the floral gradient because of tradeoffs resulting from deforestation. Indeed, adequate riparian vegetation is required to maintain (1) habitat resources (snag abundance and floral-height diversity) for riparian wildlife and (2) bank stability, contaminant and sediment filtering, shading/cooling, and input of food (detritus) and habitat (woody-debris) resources for aquatic animals (Ackerman 1993; Bottom et al. 1985; Ontario Ministry of Natural Resources 1988; Gresswell et al. 1989; Orians 1992;

Table 7. Habitat-use guilds for aquatic megainvertebrates and fishes along the riparian-floral axis, based on multivariate (varimax-factor) analysis of percent-abundance data and triplet-wise comparisons of density data. GEN = generalized and JV = juvenile. Tentative trophic guilds are given in parentheses for each taxon, i.e., drift insectivore **(DI)**, drift herbivore **(DH)**, benthic insectivore **(BI)**, or benthic omnivore **(BO)**. See Tables 1 and 5 for format and other abbreviations.

I. **Wooded**
- A. FO-SF-SH = redside shiner (*Richardsonius balteatus*) **(DI)**, chinook salmon JV (*Oncorhynchus tshawytscha*) **(DI)**, rainbow trout/steelhead (*Oncorhynchus mykiss*) **(DI)**, and Pacific lampey (*Lampetra tridentata*) **(DH)**
- B. SF = *Pteronarcys* (plecopteran) NM **(BO)**
- C. GEN-SF = chiselmouth (*Acrocheilus alutaceus*) **(BO)** and longnose dace (*Rhinichthys cataractae*) **(BI)**

II. **Unforested**
- A. GEN-SH = gomphid (dragonfly) NM **(BI)**
- B. SH = leopard dace (*Rhinichthys falcatus*) **(BI)**
- C. SH-GR = mountain whitefish (*Prospium williamsoni*) **(BI)** and largescale sucker (*Catostomus macrocheilus*) **(BO)**
- D. GEN-GR = bridgelip sucker (*Catostomus columbianus*) **(BO)**, prickly sculpin (*Cottus asper*) **(BI)**, and perlid (plecopteran) NM **(BI)**

Brewin 1996). Studies in SR, NR, and other tributaries of the Thompson River suggest that semi-forested sites are more similar to forested than unforested sites in preventing bank erosion and channel widening (Beeson and Doyle 1995; Miles 1995a, b).

The trends for habitat and biological diversity may reflect the fact that semi-forested (cottonwood–ponderosa pine) and unforested (bunchgrass–sagebrush) conditions have historically characterized stream valleys in the B.C. interior because of fire and other natural disturbances, such that native riparian and aquatic animals are adapted to semi-wooded conditions (British Columbia Environment 1991; Hooper and Savard 1991; British Columbia Ministry of Forests and British Columbia Ministry of Environment 1995). A similar importance for fire in creating semi-forested riparian conditions has been recognized in Alberta (Alke 1995; SALASAN Associates and Dovetail Consulting 1995; Brewin and Monita 1998) and in boreal Canada (Kelsall et al. 1977). Long ago, Greene (1950) suggested that thinning of riparian vegetation in cold, headwater areas could benefit downstream aquatic-invertebrate foods and trout fisheries by moderating (slightly increasing) summer water temperatures, as long as headwater deforestation is not extensive and damaging to winter cover (Bottom et al. 1985; Hassler 1987; Slaney and Martin 1997). Indeed, subsequent studies in PNW (Warren et al. 1964; Burns 1972; Lyford and Gregory 1975; Salo and Cundy 1987) and elsewhere in North America (King 1975; Grant et al. 1986; Hunt 1988; Ontario Ministry of Natural Resources 1988) have shown that removal of riparian vegetation over short stream reaches often benefit (or at least do not harm) salmonid fisheries by improving aquatic habitat (e.g., better macrophyte cover) and/or aquatic-invertebrate foods without unduly increasing summer water temperatures or sediment inputs.

Habitat and biological trends may not have been consistent across sites because of confounding factors such as urbanization, mining, irrigation, pesticide, and other human impacts, as well as differing compositions of riparian-floral species and benthic sediments (not assessed here). Therefore, examination of several biological parameters should provide a more robust assessment of human impacts (Davis and Simons 1995; Michel 1997). Ability to establish biological indicators should be improved by increasing spatiotemporal replication (examining a greater number of sites and comparing samples collected among seasons and years) and by conducting manipulative experiments (where possible, such as in northern B.C.) to compare impacted and pristine watersheds (Keeley and Walters 1994; Mellina and Hinch 1995; Tschaplinki 1996). Such efforts will require long-term, interdisciplinary research at multiple spatiotemporal scales (Fisher 1991; Vadas and Vadas 1995; Covich 1996).

The information provided in the present paper should benefit researchers and managers in western Canada, by providing solid data bases and standardized methodologies for human-impact assessment of inland watersheds. Local First Nation and other

citizens developing biological-monitoring programs in association with federal agencies in the SR (Neskonlith Indian Band, Neskonlith Fisheries Crew 1993; Crowe 1995; Culp et al. 1997), NR (Axys Environmental Consulting Ltd. 1994; Coast River Environmental Services Ltd. 1994; Nicola Valley Tribal Council 1994), and other watersheds in southern B.C. (Dovetail Consulting and Argent 1994; Department of Fisheries and Oceans 1994; Nener et al. 1997) should find the biological indicators developed here useful for evaluating the success of their riparian-restoration (tree-planting, fencing, bank-stabilization, and wetland-mitigation) programs (Vadas 1996; Broersma et al. 1995; Dovetail Consulting 1996; McPhee et al. 1996; Stavinga and MacDonald 1997). Riparian afforestation and riprapping with large rocks are expected to be more successful than other mitigation techniques for reducing bank erosion in Thompson River tributaries, although revegetation efforts are less damaging than riprapping for wetland (off-channel) and downstream habitats (Doyle 1992; Beeson and Doyle 1995; Miles 1995a,b; McPhee et al. 1996). Miles (1995a,b) suggested that riparian buffers should be 6–7 channel widths wide, i.e., 55–175 m in the middle to lower SR and 70–280 m in the middle to lower NR (cf. Department of Fisheries and Oceans 1991a, b; Vadas 1998b).

It is hoped that further ecosystem research will be done to better define riparian buffer-strip needs to protect fish and wildlife resources. At present, efforts to protect PNW riparian vegetation have been directed towards water quality and fisheries management, with much less focus on habitat needs for riparian vertebrates (Golde 1986; Raedeke 1988; Pearce 1993). For example, stream salmonids are often less sensitive to riparian deforestation than other lotic animals and riparian wildlife (Gunderson 1968; Szaro and Rinne 1988; MacDonald et al. 1991), although not invariably (Welch et al. 1977; Kauffman and Krueger 1984; Ontario Ministry of Natural Resources 1988). Indeed, some riparian vertebrates require buffer strips 100–300 m wide to maintain population viability, much wider than needed by aquatic biota (Burke and Gibbons 1995; Vadas and Newman 1998).

Because vertebrate and invertebrate taxa often differ in mobility, floral-cover preference, and thus sensitivity to deforestation (Szaro and Rinne 1988; MacDonald et al. 1991), an ecosystem focus is especially critical to assess the impacts of riparian deforestation (Szaro and Rinne 1988; Scruton et al. 1995; Tschaplinki 1996) and to improve riparian and aquatic management (Leopold 1939; Harman et al. 1984). Ecosystem research is receiving greater attention by stream ecologists in North America (Cushing 1994; Covich 1996) and natural-resource agencies in western Canada (Environment Canada 1991; British Columbia Ministry of Forests and British Columbia Ministry of Environment 1995; Fraser River Action Plan 1995; Michel 1997), including the development of biological indicators (sensitive riparian and aquatic taxa) (Gibbons et al. 1995; Landucci 1995; Landucci and Hanawa 1996; Moul et al. 1996) and strategic (integrated-watershed) planning (O'Riordan 1981; Quadra Consultants Ltd. 1995) in the Fraser River basin.

Finally, more PNW research attention needs to be directed towards the effects of riparian deforestation on larger streams with deciduous (e.g., cottonwood) buffer strips, for three reasons. First, these streams harbor aquatic and riparian faunas not present in upstream, headwater areas with coniferous riparian zones. Indeed, riparian-vertebrate biodiversity is higher downstream in PNW because water conditions are favorable (deep and stable) and floodplain vegetation is well developed (Raedeke 1988; Morgan and Lashmar 1993). Likewise, chinook salmon inhabit larger rivers than do other lotic-anadromous salmonids (Platts 1974, 1979; Allen and Hassler 1986; Sullivan et al. 1987). Second, deciduous and coniferous trees are both important in the life cycles of cavity-nesting birds (Thomas et al. 1978; Steeger et al. 1995). Third, valley vegetation protects the corridor function of watersheds, given that forested riparian corridors enhance avian migrations (Ohmart and Anderson 1982; Knopf and Samson 1994; Machtans et al. 1996) and help maintain cold temperatures (<20–25%C) needed by anadromous salmonids (Beschta et al. 1987; Walthers and Nener 1997) that spawn in headwater streams (e.g., steelhead trout and coho salmon). Hence, although salmonids in North America are generally located in northern regions with coniferous vegetation (Brinson et al. 1981) and riparian management for PNW salmonids are focusing on replanting conifers in headwater streams (Berg 1995), downstream riparian rehabilitation with willow and other deciduous vegetation (as in the SR and NR valleys) is also critical for protecting fish and wildlife. Clearly, a watershed approach is needed to effectively manage natural resources in and near PNW rivers.

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An Ecosystem Diagnostic Tool for Adaptive Management for Fisheries Resources

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Abstract

This work was performed in the Grande Ronde Watershed, State of Oregon, U.S.A. in September 1994 through June 1995. The project was performed to address the complex question of how to develop, prioritize, and implement salmon habitat restoration projects for Pacific anadromous salmon that recognizes the importance of an ecosystem perspective. An ecosystem diagnostic tool and six-step restoration planning process are described. The approach includes a comparison of historic and current habitat complexity and connectivity and intrapopulation life-history diversity. This report illustrates a comparison of the historical and present habitat conditions by attribute and reach, and their relative effects upon the in-basin productivity by life cycle stage of the Snake River Spring Chinook, Oncorhynchus tshawytscha. Fourteen environmental attributes were analyzed, and a composite productivity index was developed for each of four life history trajectories. The analysis showed that present conditions offer a narrow window for successful completion of the full in-basin life cycle of the spring chinook. In the Upper Grande Ronde, present habitat productivity would have to increase by 3.6 to 20.0 times to increase smolts/spawners to 100. In Catherine Creek, habitat productivity would have to increase by 2.3 to 2.7 times to achieve the same results.

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Introduction

This progress report presents initial results of a technical analysis undertaken to provide assistance in the formulation of actions aimed at improving the sustainability of natural resources and related benefits within the Grande Ronde watershed. The analysis was performed using a methodology referred to as Ecosystem Diagnosis and Treatment (EDT) (Lichatowich et al. 1995), which was adapted for specific application to the Grande Ronde Basin. Results reported herein primarily pertain to the Upper Grande Ronde system (upstream of Wallowa River).

The analysis focuses on spring chinook salmon, which serves as a diagnostic species in the assessment of the condition of the watershed for sustainability of its resources and related societal values. This study assumes that humans and their values are integral parts of an ecosystem, and that human communities within the Grande Ronde Basin desire a healthy watershed–one that can sustain natural resources as well as economic and social values for future generations.

The EDT methodology requires that a clear diagnosis be formulated based on existing knowledge. The diagnosis is used to guide the development of rational actions, which can take many forms, such as habitat improvement projects, modifying land and water use activities, and intervention using artificial production techniques (i.e., supplementation). The major intent of this report is to illustrate how this diagnosis is formed for the Upper Grande Ronde system.

Objectives

The purpose of the Grande Ronde Ecosystem Diagnosis and Treatment Project (Project) was to provide technical assistance to the Board in their effort to plan and implement watershed recovery projects.

Specific Project objectives were to develop and describe:

1) a science-based planning process that effectively incorporates local values and objectives;
2) scientifically sound methods for: a) identifying factors that inhibit achievement of sustainable watershed recovery objectives, b) prescribing potential recovery actions, c) prioritizing actions, d) analyzing trade-offs between actions, and e) monitoring and evaluation to manage risks and improve future plans.

This abbreviated report focuses on the Patient and Template analyses of habitat productivity. This information is applied to a potential prioritization for habitat protection, restoration, and enhancement.

Methods

Perform Analysis and Diagnosis

A generalized approach for comparing existing and desired conditions is called the Patient–Template Analysis (PTA) (Lichatowich et al. 1995). This approach uses medical analogies to compare existing conditions of the target populations and their habitats (Patient analysis) with hypothetical healthy conditions (Template analysis) to form a diagnosis of the subject's status and arrive at a set of prescribed treatments.

Patient–Template Analysis

The PTA procedures are described in the following sequence of steps: system organization, assessment measures, summarization and analysis, and graphical display.

System organization

All of the information used in or produced from the PTA needs to be considered within spatial-temporal scales consistent with the range of possible life histories of the diagnostic species. This requires that spatial-temporal scales be defined accordingly.

The Grande Ronde watershed is divided into separate units on the basis of the natural stream drainage system, as illustrated in the schematic shown in Figure 1. The schematic shows part of the drainage system broken into tributary and mainstem units used in the analysis. The units are not equal in size. Units are delineated by the connectivity of the drainage along stream channels using the EPA's stream reach numbering system as expanded by the Northwest Power Planning Council. Unit delineation in Figure 1 shows the level of spatial organization selected for analyzing spring chinook. Other scales may be required for other diagnostic species.

Time also needs to be divided into relevant units. For salmon species time is delineated within a calendar year by month and statistical week. This scale is well suited for salmon species. Life stages of salmon can be identified as distinct developmental phases that have different environmental requirements. These stages can be defined by month or statistical week, as illustrated for seven life stages of spring chinook salmon with the Grande Ronde Basin (Table 1).

Much of the information used in formulating the diagnosis is displayed in a format that captures both

Table 1. Definition of spring chinook life stages within the Grande Ronde Basin and corresponding time periods

Life stage	Description	Months	Statistical weeks
Pre-spawning adult	Upstream adult migration and holding prior to spawning.	April–August	15–33
Spawning	Spawning period, including establishment and defense of redd sites.	August–September	33–38
Incubation	Egg deposition to fry emergence.	August–April	33–70
Fry colonization	Fry emergence until establishment of summer rearing locations.	April–June	61–71
Summer rearing	After colonization ceases when fish are largely stationary and activities are mainly directed at feeding and growth (large fish may outmigrate near end of period).	June–September	71–95
Fall redistribution and overwintering	Beginning with drop in temperature in early fall until the onset of yearling smolt migration at the end of winter.	September–March	96–114
Smolt to smolt	Onset of seaward migration to departure from the Grande Ronde Basin.	April–June	114–125

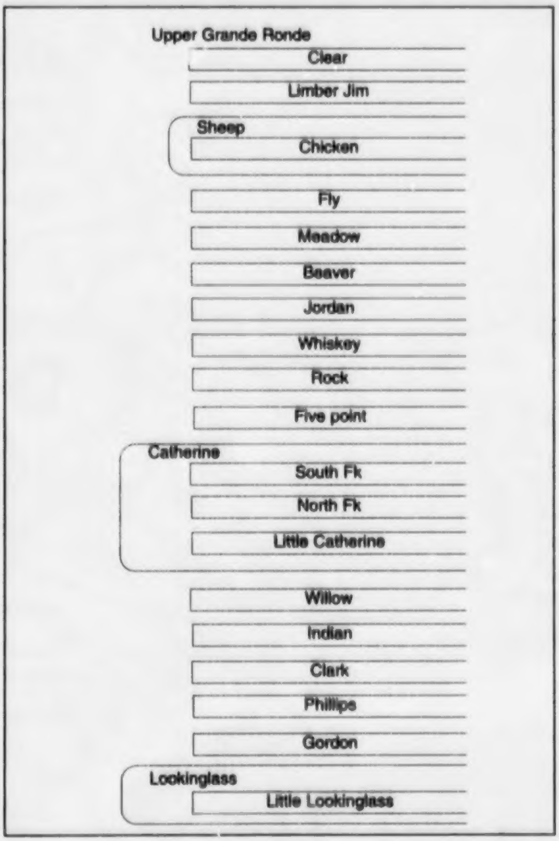

Figure 1. Spatial scale used for the Grande Ronde Basin framework.

the space and time dimensions, as shown in Figure 2 for the mainstem Grande Ronde. The figure illustrates format only and contains no information. This format is used as a device to help visualize patterns of survival and the relative strengths of mortality factors operating on the population in time and space. This format is particularly effective at showing how conditions that affect the sustainability of a population can vary dramatically within these dimensions.

Assessment measures

Four measures are employed to assess the effects of environmental attributes on population performance: 1) relative productivity (or survival), 2) relative effect of environmental quality on productivity, 3) total quantity of habitat, and 4) relative quantity of key habitat (or proportion of total). The combination and interaction of these measures describe population performance in relation to the unique set of environmental quality and quantity attributes of a watershed. Each measure, except one, is assessed in a relative manner (i.e., as an index rather than absolute estimate). This simplifies the analysis. All measures are assessed for each life stage of the diagnostic species within each geographic unit.

Relative productivity

This measure describes that element of performance referred to as productivity. It addresses

mortality or losses due strictly to density-independent processes. Participants at the working session assess spring chinook productivity based on their knowledge of the environmental requirements considered optimal for the species and of conditions within the geographic units.

The measure is scored on a scale of 0–1, where a value of 0 represents no survival and 1 represents optimal survival conditions (ignoring density effects) for the diagnostic species. If, for example, a river reach is given a value of 1 for one life stage, say egg incubation, this would mean that conditions in that reach are considered optimal for egg survival. Therefore, survival in the absence of density effects would be the highest possible for this life stage under natural conditions. If the reach is given a score of 0.5, then survival is expected to be equivalent to 50% of the highest possible under natural conditions. If a score of 0 is given, then no survival is expected for the stage.

Relative quality of environmental attributes

This measure explains, or justifies, the productivity scores by identifying the relative contributions of a range of environmental quality attributes to those scores. The measure describes how participants in the working sessions rate the effects of environmental quality conditions within each geographic unit on life stage productivity.

Fifteen descriptive attributes of environmental quality for spring chinook were identified. These include channel stability, flow, habitat type, diversity, sediment load, temperature, riparian condition, predators, chemicals, competitors, obstructions, withdrawals, nutrient load, oxygen, pathogens and other. All of these attributes are known to affect the density-independent survival of salmonids at one or more of the life stages in freshwater. Three of these attributes (competitors, predators, and pathogens) are biotic factors representing non-diagnostic species and their effects on the diagnostic population; they are treated as part of the environment affecting the diagnostic species.

Participants at the working session score each quality attribute by identifying its relative contribution to the productivity scores given. Attributes are scored on a scale of 0–4, where 0 indicates no contribution to downgrading survival (from optimal) and where 4 indicates a lethal effect. The range is lethal effect = 4, high effect = 3, moderate effect = 2, low effect = 1 and no effect = 0. For example, if relative productivity was scored at a value of 1, indicating optimal conditions for survival are present, then all quality attributes would be scored 0 (i.e., no deleterious attribute effects). If relative productivity had been scored 0.5, indicating less than optimal conditions are present, then at least one or more attributes would likely have been scored a 2 or 3 indicating a moderate or high effect on survival. If productivity had been scored 0, indicating no survival is expected, then at least one attribute would have to be scored at a value of 4 for a lethal effect.

Quantity of habitat

This measure quantifies the total amount of habitat available to be used by the diagnostic species within each geographic unit in each life stage, including areas that may not be highly preferred or utilized. For spring chinook, total available habitat would consist of the total amount of stream area available to be used in each geographic unit and life stage. Stream area is computed as the product of stream length and average width (wetted area) by time period. The data needed for these computations are obtained from stream habitat databases. This measure is used in conjunction with the relative quantity of key habitat (see next section) to analyze the distribution of habitat capacity, one of the three performance elements.

Relative quantity of key habitat

This measure quantifies the amount of key habitat relative to the total amount available within each geographic unit in each life stage. Habitat requirements and preferences differ by species and often by life stage for those species. The key habitat measure is used as a way of examining habitat capacity in the diagnosis.

Key habitat is that component of the total habitat available to a species that is strongly preferred, or needed, during a life stage. These include: Adult: Large, deep pools with sufficient connecting flow for adult migration. Spawning: Riffles containing a mixture of gravel and cobble sizes with flow of sufficient depth for spawning activity. Incubation: Riffles as described for spawning with sufficient flow for egg and alevin development. Fry colonization: Shallow and relatively slow velocity areas within stream channel, often associated with stream margins and in relatively low gradient reaches. Summer rearing: Pool type habitat associated with relatively low gradient stream channel reaches (usually not in backwaters nor slow eddies). Fall redistribution/overwintering: Areas containing structural complexity (wood matrices, brush, or large cobbles) within flowing channel, not usually in swift or higher gradient

Table 2. Relative quantity of key habitat for spring chinook within stream reaches of the Grande Ronde Basin

Relative quantity of key habitat	Score	All stages except smolt to smolt[a]	Smolt to smolt life stage
Exceptionally high	4	>50% of stream area	Superabundance of needed flow
High	3	>25% and <50% of stream area	Migration may be affected slightly
Low	2	>5% and <25% of stream area	Migration affected noticeably by reduced flow
Scarce	1	>0% and <5% of stream area	Migration very difficult due to low flow
None	0	0% of stream area	Channel is dry

[a] Stream area being referred to during fry colonization is the area along stream margins.

reaches; off-channel areas (ponds, oxbows, etc.). Smolt to smolt: Sufficient flow for free movement of smolts downstream. For example, salmon require a stream to live in, but they also require riffles containing a certain sized substrate for spawning and reproduction. These spawning riffles are referred to as 'key habitat' during both the spawning and egg incubation life stages. In this example, the measure would indicate the percentage of stream environment within a geographic unit that consists of spawning riffles suitable for chinook salmon at the appropriate time. The measure says nothing, however, about the relative quality of spawning riffles for egg survival; that is described through the productivity measure.

The relative amounts of key habitat are determined according to five categories of availability using scores of 0–4 (shown for spring chinook in Table 2). Here, a score of 0 indicates that no key habitat is present, whereas a value of 4 indicates that it is superabundant relative to the total habitat present. Use of categories of availability in this manner facilitates acquisition of information.

Summarization and analysis

All relevant quantitative information is stored in a computerized database (MS Access 2.0). Related descriptive comments obtained through the work sessions are placed in the same database.

The database also serves as the primary device for documenting assumptions, thereby creating a permanent record of the analytical process to be used in tracking the logic through the project, formulating hypotheses, and identifying monitoring needs.

The data are formatted into the appropriate spatial and temporal scales using a set of analytical steps coded in MS Excel 5.0. These steps are used to build data lists for use with SYSTAT software, required for producing many of the graphical displays.

Graphical display

Visual displays of information are the primary means of comparing conditions between Patient and Template for the diagnostic species. They provide visualization of patterns in space and time dimensions that are relevant to the overall condition of the resource under inspection. A set of standard formats, such as those depicted in Figure 2, facilitate visualization and subsequent analysis.

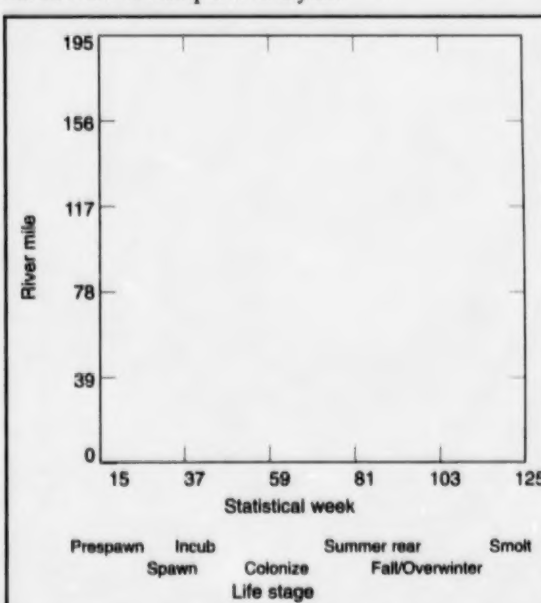

Figure 2. Spatial–temporal format for displaying information for spring chinook along the mainstem Grande Ronde River.

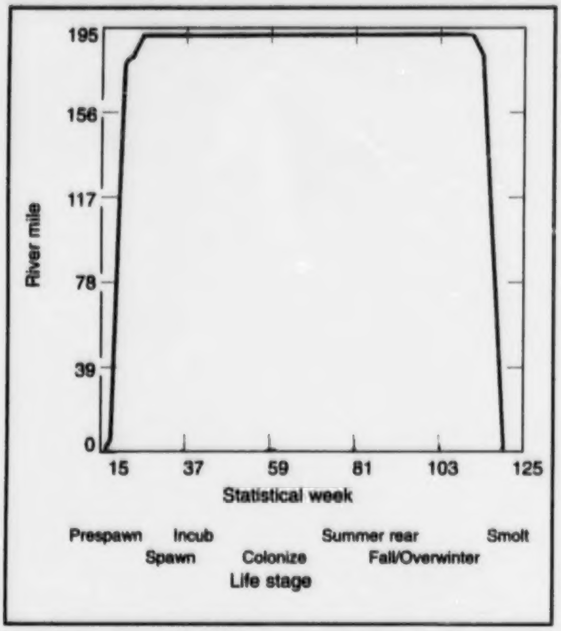

Figure 3. Example life history trajectory in the Grande Ronde Basin.

All graphical displays are produced with SYSTAT 5.0 and Excel 5.0. Charts plotting information in space and time are made using a set of routines designed in SYSTAT using contour and 3-D surface plotting functions.

Diagnosis

The diagnosis is a determination of the general condition of the diagnostic species to sustain itself and related societal values, given past and present resource uses and environmental change. The diagnosis is made by viewing and analyzing the information assembled in the PTA from a life history perspective.

Life history trajectories

In performing the diagnosis, consideration is given to the likelihood that conditions for sustainability may vary greatly within the watershed, both in geographic space and time. This poses an analytical challenge because there are a myriad of possible sets of conditions that different members of a population like salmon can experience throughout a watershed. Habitat conditions for species as migratory as salmon are likely to be highly heterogeneous within a geographic area the size of the Grande Ronde watershed.

This challenge is addressed by defining sample pathways, or trajectories, that members of the diagnostic species can follow through the watershed, both in space and time. This procedure, or analytical probe, is a way of sampling the possible sets of conditions that animals can encounter in completing their life cycles.

An example trajectory is illustrated in Figure 3, where one possible path of a spawner and its progeny is traced in space and time within the watershed. The path begins in the lower right corner of the chart with the entry of an adult migrant salmon into the Grande Ronde River from the Snake River. In this case the fish enters in statistical week 16, or about mid-April. The path continues upstream, charting the progress of the migrant adult to the spawning grounds in the Upper Grande Ronde River. At spawning, the path then represents progeny of the spawner, beginning as eggs and continuing through subsequent life stages until seaward migration as smolts.

A single trajectory can be defined in this manner such that it is consistent with a known life history pattern. Many trajectories differing only slightly from the one could potentially be defined so that the entire bundle of pathways would be representative of the life history pattern. A different life history pattern would be characterized by a different set of trajectories.

Series of trajectories can be defined in this manner, consistent with known life history patterns for the diagnostic species, facilitating analysis and comparison of the unique combinations of conditions and associated performance characteristics along each defined path. The conditions associated with major and minor life history patterns can thereby be characterized and compared.

This analytical device is one of the most significant aspects of this diagnostic approach. It facilitates analysis of performance across the watershed and provides the means for linking all other life stages that occur outside the drainage. In doing so, a comparison can be made of the sustainability of complete life cycles. Factors affecting mortality along the full life cycle can then be identified and analyzed.

Productivity index

An index of productivity is computed as a way of analytically comparing the composite productivity between sample life history trajectories. The composite productivity (after Moussalli and Hilborn 1986 and Hilborn and Walters 1992) is the overall

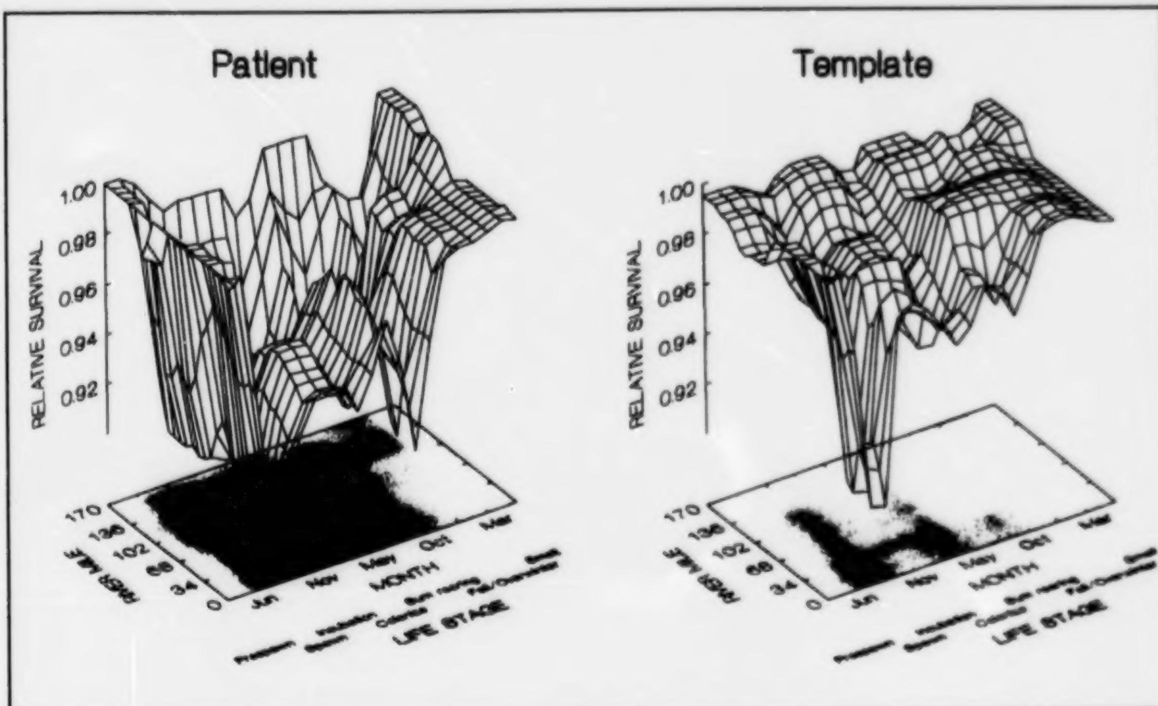

Figure 4. Grande Ronde River relative survival.

productivity for an entire life history trajectory. Productivity can be expressed as a density-independent survival rate; hence composite productivity can be expressed as a survival rate that encompasses all life stages traversed by a life history trajectory. The values of relative productivity described earlier in deriving the index are used here.

To compute the index, a profile of productivity for a hypothetical population existing under optimal environmental conditions at all life stages is first described. Productivity, or density-independent survival, would be the highest that could be theoretically attained in each life stage.

These values of productivity would be approximately equivalent to real survival rates for a population living in optimal habitat conditions at very low population densities, where density-dependent losses would be negligible.

A set of values for spring chinook productivity under optimal conditions was obtained through consultation with Dr. Ted Bjornn and the review of pertinent literature. Selected productivity values by life stage are: Adult migration/holding—0.95, Spawning—0.95, Incubation—0.70, Fry colonization—0.80, Summer rearing—0.70, Fall redistribution/overwintering—0.70, and Smolt to smolt—0.95.

The composite productivity across all these life stages is simply all life-stage rates multiplied together (see Hilborn and Walters 1992, page 285):

$$P_n = \prod_{i=1}^{n} p_i$$

where P_n is the composite of productivities through the nth life stage and p_i is the productivity in stage i expressed as the density-independent survival rate for that stage.

Therefore, our estimate of a productivity standard across all seven stages under optimal conditions is 0.25.

As a survival rate that encompasses all of the life stages shown above, this can be thought of as the proportion of eggs per adult entering the river that eventually survive to depart the river as smolt.

The productivity value, or survival rate, can also be expressed in terms of the number of progeny, or

smolts, produced per adult salmon entering the river. It is assumed, for the purpose of computing index, that an adult spring chinook salmon in the Grande Ronde River has, on the average, 2700 eggs. It is also assumed that the sex ratio averages 1:1 and that these values are representative of both Template and Patient characteristics.

The productivity profile is then converted to a weekly time scale to enable us to define p values along the trajectory pathway. This is done by taking the mth root of p_i:

$$P_w = \sqrt[m]{p_i}$$

where P_w is the productivity in week w, and m is the number of weeks defining the time period associated with that life stage.

The Productivity Index (PI), expressed as a rate, for n life stages is then calculated:

$$PI_n = \prod_{\substack{w\\r}} Rp_{r,w} \, P_w$$

where PI_n is calculated as the product of the productivities for all reach r and week w units traversed by the trajectory and $Rp_{r,w}$ is the relative productivity associated with stream reach r and week w (as described earlier).

The Productivity Index can also be expressed as a number of smolts per pre-spawning adult by multiplying the rate by the number of eggs per adult entering the river. Results here are presented using this form of the index.

The Productivity Index values are then used to compare the health (i.e., sustainability) of different life history trajectories and the patterns they represent. Values are applied in two ways. First, the index values indicate relative differences in the sustainability of various life history patterns. The values show how much variability in productivity exists within life history patterns (between trajectories) and between life history patterns. Productivity indices for similar trajectories in the Patient and Template are also compared.

Second, values are used in conjunction with information on smolt-to-adult survival rates as a measure of the composite productivity for the full life cycle. For example, if within basin productivity for a trajectory is 100 smolts per adult and smolt-to-adult survival is 2%, then the composite productivity for the entire life cycle would be 2 returning adults per parent spawner. In the absence of population density effects or environmental fluctuations, a life history with this level of productivity would be sustained. In contrast, if within-basin productivity is 40 smolts per adult, then the composite productivity for the life cycle would be 0.8 returning adults per parent spawner. In this case, the fish following this life history path would be a drain to sustainability.

This procedure is meant to help reveal general patterns of condition and their relative importance to one another. The computed indices can provide insights about the magnitude and extent of conditions affecting sustainability. Differences between Patient and Template conditions suggest historic changes in productivity. Results should not be used for predictive purposes. Ecological processes are too variable and our understanding about them is too limited for that purpose.

Summary determination

The final step in the diagnosis consists of a summary determination of the general condition of the diagnostic species and the relative contributions of factors affecting the species. The determination is made within the context of program objectives. Large amounts of information must first be viewed at small scales defined by individual trajectories and at much larger scales that show broad patterns across the landscape.

The summary determination is where the scientists doing the analysis integrate all the information across these scales into clear concise statements that summarize the diagnosis. The determination needs to be supported by and consistent with the analytical results. These summary statements, combined with key visual displays, are the basis of communicating the diagnosis to decision makers.

It may be useful, even necessary, to formulate more than one plausible diagnosis to help identify information needs for future work.

Treatment Identification

The purpose of the treatment identification step in the planning process is to assemble a collection of candidate actions. Proposed actions can come from many sources, from individuals, organizations, and agencies. The treatment identification procedure should assure that among the collection of alternatives are some that are based upon the diagnosis.

The procedure for identifying actions consistent with the diagnosis involves first formulating one or more basin-wide strategies. A strategy sets an overall

Table 3. Productivity index values (in smolts per spawner) for different trajectories within the primary and secondary life history patterns of spring chinook in the Upper Grande Ronde mainstem

Life history pattern	Trajectory description	Productivity (smolts/spawners)
Upper Grande Ronde - Primary (spawn, rear, and overwinter)	Spawns upstream of Clear Creek	185
	Spawns upstream of Limber Jim Creek	28
	Spawns upstream of Sheep Creek	6
	Spawns below Sheep Creek	5
Upper Grande Ronde - Secondary (spawn, rear, overwinter in valley)	Spawns upstream of Clear Creek	208
	Spawns upstream of Limber Jim Creek	32
	Spawns upstream of Sheep Creek	6
	Spawns below Sheep Creek	5

direction to guide the development of watershed improvement actions. Basin-wide strategies should be framed upon principles of watershed dynamics, ecosystem function, and conservation biology. These principles can be simply captured in one general principle using a life history perspective for the diagnostic species.

In the simplest terms, the principle calls for setting the following priorities: first, maintaining; second, improving; and third, restoring. The conditions (or health) of existing life history patterns for the diagnostic species are the criteria for establishing strategic priorities. The rationale for this principle is that it is more prudent to maintain and make secure existing life history patterns before attempting to restore or recover patterns that have been disrupted through past changes to the watershed. At existing levels of production of Grande Ronde spring chinook, it is reasonable to apply this principle solely to the productivity of existing life history patterns.

Results and Discussion

The initial results for the Upper Grande Ronde watershed are presented in the following three sections: 1) Patient–Template Analysis; 2) Diagnosis; and 3) Treatment Identification.

Patient–Template Analysis

The PTA for the Upper Grande Ronde system is based on an assessment of conditions in the entire drainage upstream of Wallowa River at RM (River Mile) 80 and for the mainstem Grande Ronde River from the Wallowa River to the Snake River. Inclusion of the entire mainstem river enables us to consider all life stages for the diagnostic species within the entire Grande Ronde watershed. Spring chinook salmon produced upstream of Wallowa River are dependent on the lower river to complete their life cycles.

Relative productivity

Relative productivity is a measure of density-independent survival, where a value of 1 is equivalent to the highest possible survival rate under optimal conditions in nature. It does not, as a relative measure, identify what actual survival rates (ignoring density effects) would be.

We found that major changes have likely occurred in spring chinook productivity within the Grande Ronde watershed between historic and current conditions. Along the mainstem Grande Ronde River, from its mouth to the headwater reach near RM 195, productivity appears to have declined substantially for portions or all of each life stage that occurs in these waters (Fig. 4).

While relative productivity for the historic Template analysis appears comparatively consistent across the space and time landscape, it varies dramatically under current conditions (Fig. 4).

In general, relative productivity appears to be highest today for adult migrants moving through

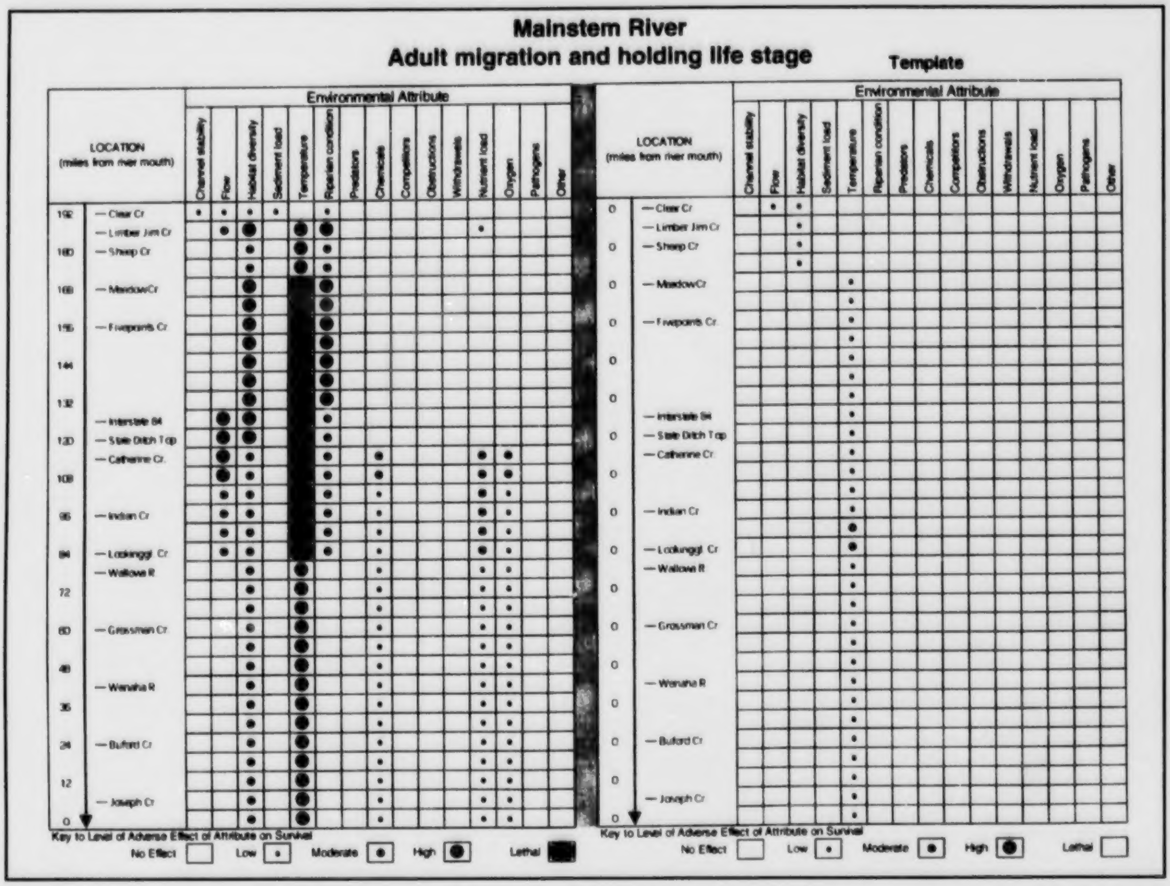

Figure 5. Life-stage specific summaries in the mainstem Grande Ronde River of relative effects of environmental quality attributes on spring chinook productivity (survival); adult migration stage.

the system early (i.e., in April and May) and for smolts departing the system in spring of their second year of life. Relative productivity appears to be particularly low for fry colonization, summer rearing, and overwintering in large segments of the mainstem Grande Ronde River today.

It should be noted that the values are expressed as weekly rates of relative survival. Low values approach 0.9, which may not seem particularly low at first consideration. These values are multiplicative, however, meaning that the relative survival for all life stages combined is the product of an entire string of weekly rates. For example, a weekly rate of 0.95 across 52 weeks results in a composite rate for the period of 0.07 (i.e., 0.95^{52}).

Environmental quality attributes

The productivity, or density-independent survival, of a population is a function of environmental quality. The relative contributions of 15 attributes of environmental quality to productivity within the watershed is displayed in a consumer's report style format for comparing the relative importance of different attributes between geographic units. This format is used for comparing conditions between mainstem river geographic units and also between tributary sub-drainages. Only the adult migration and holding life stage is displayed in Figure 5.

Major changes have apparently occurred for most environmental quality attributes affecting spring chinook productivity within large segments of the watershed (Fig. 5). Of the 15 attributes, we judged the effects of channel stability, flow, habitat type diversity, sediment load, temperature, riparian condition, and predators to have generally increased the most. Effects of these changes have not occurred uniformly throughout the watershed nor across the life stages of spring chinook.

Table 4. Procedures for developing actions consistent with strategic priorities

Step	Procedure
1	Identify primary and secondary life history patterns (based on the life history analysis).
2	Identify where and when productivity can potentially be improved, giving particular attention to areas used for longer time periods (based on life history analysis and Productivity Index).
3	Identify environmental quality attributes that need to be addressed to improve productivity along the corresponding pathways.
4	Identify actions that would be required to change the appropriate environmental attributes (from preceding step) to a desired level.

Overall, changes in water temperature between historic and existing conditions appear to have had the greatest contribution in reducing spring chinook productivity. Changes in this attribute have likely affected all life stages of spring chinook within the watershed. Within the mainstem Grande Ronde River, for example, increased water temperatures have likely occurred during late spring, summer, and early fall months, increasing mortality during adult migration and holding, spawning, egg incubation (during initial weeks), and summer rearing life stages. In addition, temperatures have apparently decreased in some river reaches during winter, thereby increasing mortality during egg incubation and overwintering life stages in those areas. These changes in temperature are assumed to have occurred as a result of widespread alterations in riparian conditions within the upper watershed and its tributary sub-drainages and due to changes in snow and ground water retention in the uplands of certain sub-drainages. Similar changes appear to have occurred in the temperature regimes of many other watersheds inhabited by spring chinook east of the Cascade crest (e.g., Lichatowich and Mobrand 1995).

Changes in the diversity of in-stream habitat appear to have been extensive throughout the areas examined. Effects of changes in this attribute are likely widespread along the mainstem river during the adult migration, fry colonization, summer rearing, and overwinter stages. These changes are largely explained by major alterations that have occurred in in-stream structure (e.g., large wood), channel morphometry, and the riparian corridor (see McIntosh 1992, Huntington 1994, and McIntosh et al. 1994).

Alterations in the sediment load of the mainstem river appear to have reduced productivity of spring chinook throughout the length of the river. The effect was judged to be strongest during the egg incubation stage, followed by juvenile overwintering. Sources of sediment in the upper basin appear to be scattered, due to extensive land use activities that have occurred through time (McIntosh 1992). One recent event, a "one–two punch" consisting of a large fire followed by a major flood, occurred in 1989 (United States Forest Service 1994).That event appears to have resulted in a major recruitment of sediment to the upper river.

Diagnosis

The diagnosis is a determination, based on deductive analysis, of the general condition of the diagnostic species to sustain itself and related societal values as a result of past and present modes of resource use and environmental change. It includes an assessment of the relative contributions of the various factors affecting the condition of the diagnostic species. To complete the diagnosis, we examined the information presented thus far from a life history perspective. This perspective includes both the diversity of possible life histories and the necessity to view life history over the full life cycle. Pattern 1 as depicted in Figure 3 can be described as follows: Pattern description: Adults enter river early, move quickly to mainstem headwaters; progeny rear and overwinter in that vicinity, then emigrate seaward as yearling smolts. Trajectory description: Spawn, rear, and overwinter in the mainstem near Clear Creek. Three other patterns were examined but are not displayed here.

Life history trajectory

Four generalized life history patterns were selected to sample the range of Productivity Index values in the upper basin. Pattern 1 is represented by

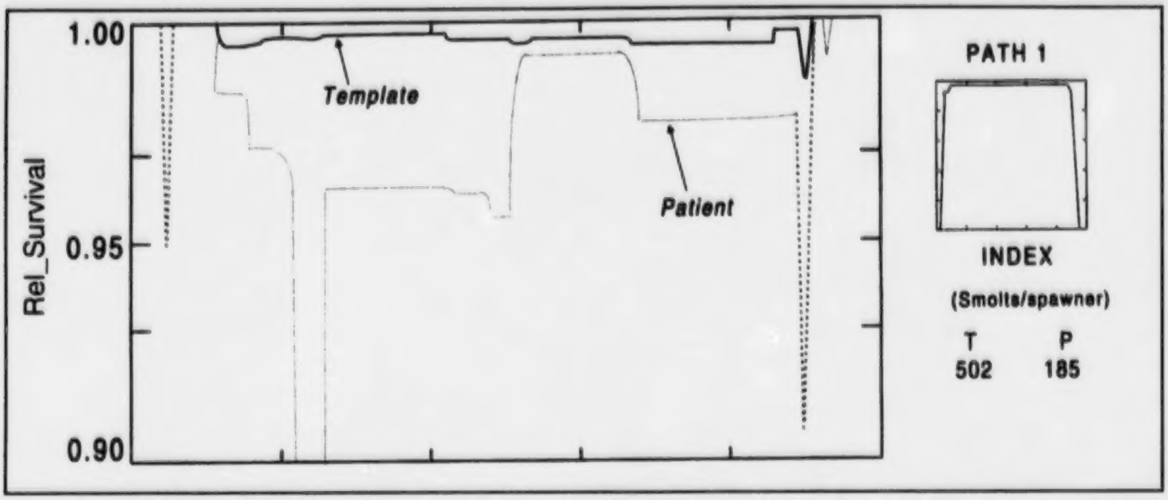

Figure 6. Relative productivities through time (by week) for life history pattern 1 (see Fig. 3) and associated Productivity Index values (smolts per spawner).

Table 5. Scale of improvement needed (as multipliers) to increase spring chinook productivity (as smolts per spawner) to levels shown for primary and secondary life history patterns in the Upper Grande Ronde Basin

Life history pattern	Trajectory description	Productivity (smolts/spawner)	Scaling factors needed to increase productivity to 50	100	150	200
Upper Grande Ronde - Primary (spawn, rear, and overwinter)	Spawns upstream of Clear Creek	185				1.1
	Spawns upstream of Limber Jim Creek	28	1.8	3.6	5.4	7.1
	Spawns upstream of Sheep Creek	6	8.3	16.7	25.0	33.3
	Spawns upstream of Sheep Creek	5	10.0	20.0	30.0	40.0
Upper Grande Ronde - Secondary (spawn, rear, overwinter in valley)	Spawns upstream of Clear Creek	208				
	Spawns upstream of Clear Creek	32	1.6	3.1	4.7	6.3
	Spawns upstream of Sheep Creek	6	8.3	16.7	25.0	33.3
	Spawns upstream of Sheep Creek	5	10.0	20.0	30.0	40.0

the trajectory (or pathway) in Figure 3. This pattern was identified based on field studies in the watershed (Keefe et al. 1994).

Pattern 1 is the primary life history pattern existing within the upper basin today. Most fish that spawn in Catherine Creek today follow a similar pattern. Studies indicate that approximately 80% of the existing smolt production in the upper basin follow this pattern. Pattern 2 is identical to pattern 1 but progeny emigrate to the Grande Ronde valley for overwintering, then emigrate seaward as yearling smolts. Pattern 3 is identical to pattern 1 until fry colonization, when fry move downstream to above La Grande for rearing and overwintering, then emigrate seaward as yearling smolts. Pattern 4 is identical to pattern 3 until late summer when juveniles emigrate seaward as subyearling smolts.

Productivity indices

The index suggests that productivity has declined sharply over the past century. The range of index values expressed as smolts per spawner for

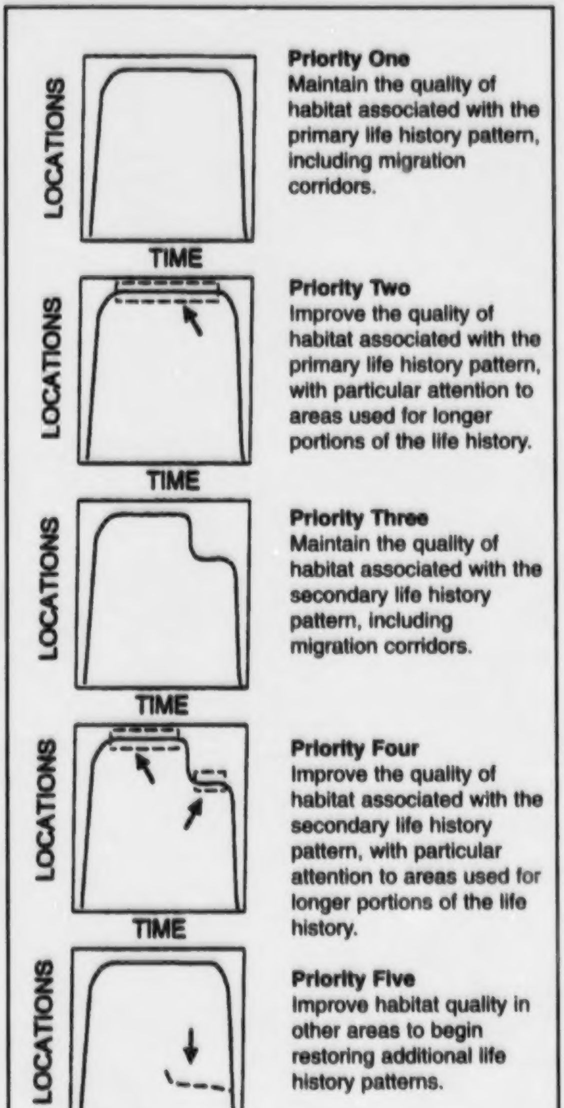

Figure 7. Strategic priorities for restoration of spring chinook in the Upper Grande Ronde.

Template analysis (T) and Patient analysis (P) for the four sample life history trajectories was: Pattern 1, T = 502, P = 185; Pattern 2, T = 577, P = 32; Pattern 3, T = 518, P = 0; and Pattern 4 T = 862, P = 0, respectively. This information is displayed graphically for Pattern 1 in Figure 6.

Under existing conditions, productivity values associated with the four patterns differ dramatically. Productivity of the primary life history Pattern 1 is highest, followed by the value for the secondary pattern. The values for the other trajectories show these patterns as drains to population productivity.

The index suggests that the most productive pattern under Template analysis conditions was number four. By migrating seaward before winter, survival from egg to smolt would have been higher relative to the other patterns.

Productivity also appears to vary substantially within these generalized life history patterns (Table 3). Trajectories following somewhat different paths through space and time, but maintaining the same basic patterns, show widely varied productivities. Table 3 compares multiple trajectories for the primary and secondary life history patterns for the upper mainstem Grande Ronde River.

Productivity index values for different life histories within the watershed can be combined with smolt to adult survival rates to assess productivity across the entire life cycle. The analysis of survival rates outside the basin is incomplete; hence results presented here are for illustrative purposes. Smolt to adult survival rates generally vary from less than 0.5% to 2% (Cramer and Neeley 1993).

Treatment Identification

A proposed strategy for guiding the development of watershed improvement actions was formulated based on the diagnosis and the strategic set of priorities. It is specific to the area of the watershed covered by this report. The strategy is portrayed visually in Figure 7 for the area encompassing the mainstem Grande Ronde River.

Actions can be developed following the steps outlined in Table 4, using information presented in Figures 3–6 and Table 2.

Implications

Spring chinook abundance has declined sharply in the Grande Ronde Basin over the past century. Relatively small numbers persist, mainly produced from spawners that utilize the upper reaches of the mainstem river and its major tributaries.

The cause of the decline is a severe reduction from historic levels in the composite productivity of the population across its entire life cycle. Productivities (survival rates) have declined both within and outside the watershed. Reduced productivities have occurred for most or all life history patterns, resulting in the complete loss of some. Lichatowich and Mobrand (1995) reached the same conclusion for other chinook populations in the region.

Outside the basin, factors affecting survival consist of passage-related problems associated with the hydroelectric system, harvest, interactions with hatchery fish, and a decline in natural marine survival (Lichatowich and Mobrand 1995; National Marine Fisheries Services 1995).

Within the Upper Grande Ronde Basin, loss of productivity appears to have been widespread. Life history patterns that persist in this area of the basin appear to be almost entirely dependent on conditions in the extreme upper reaches of the river. A similar situation exists for fish produced in Catherine Creek. Therefore, the continued sustainability of these life history patterns appears at the present time to depend on maintaining productivity as high as possible within these headwater reaches.

Loss of productivity within the watershed can be linked to past alterations of habitat quality. These changes have affected survival of most life stages of spring chinook in the upper basin.

Habitat quality upon which this species depends falls off sharply as one moves away from the headwater reaches (Fig. 4). The attributes of habitat quality that have contributed the most to loss of productivity in the upper basin are water temperature, habitat diversity, riparian condition, sediment load, and channel stability, though other attributes have also contributed (Table 2).

The composite productivity, hence sustainability, of the population can be improved by increasing productivity within the basin, outside the basin, or both. A prudent approach should involve measures both outside and inside the basin.

Table 5 illustrates the scale of improvements that may be needed within the basin to increase productivities to various levels. In general, a minimum productivity of 100 smolts per spawner would be needed to sustain a single life history trajectory if smolt to adult survival is 1%. However, the progeny of a group of spawners that utilize a single river reach will ultimately follow many trajectories as they grow and disperse, as well as several life history patterns. Some of those patterns will be a productivity drain to this subpopulation; therefore some patterns need a productivity much greater than 100 to replace the parent spawners. The table should be used only as a very rough guide for considering the magnitude of improvements that may be required.

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Watershed Analysis as a Tool for Landscape Management, Monitoring, and Restoration

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Abstract

Watershed analysis in Washington State was developed by constituents of the Timber Fish and Wildlife (TFW) agreement (Native American tribes, state agencies, environmental groups, and the timber industry) to address the cumulative effects of forest practices on fish and water resources. Timber Fish and Wildlife watershed analysis is unique in its scientific rigor and potential ability to affect land management / restoration decisions because of close interaction between scientists and policy makers. The watershed analysis process consists of three components: resource assessment, prescriptions and monitoring. An interdisciplinary team of scientists with expertise in geology, hydrology, fish biology, soils, and forest ecology assesses physical and biological components of the watershed. Watersheds range in size from 100 to 200 km^2 and take approximately 2 to 3 months to evaluate. Present and historic land management activities (e.g., timber harvest, grazing) are evaluated in the context of natural disturbance processes (e.g., floods, fire). The evaluation relies primarily on aerial photographs, monitoring data, landform maps, and geographic information systems, with some additional field work. The team delineates specific areas of the landscape that are sensitive to management practices. Land managers then work with scientists to develop options or prescriptions for operating in sensitive areas. The direct connection between the scientists and land managers ensures that information generated by the assessment team is at a scale appropriate for guiding management decisions in the field. Finally, monitoring and restoration approaches are discussed in the context of data generated from watershed analysis.

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Introduction

Strategies for protecting aquatic resources while allowing land management activities must be sufficiently flexible to accommodate landscape variability in the Pacific Northwest (Forest Ecosystem Management Assessment Team 1993). Methodologies for conducting watershed analysis in forested watersheds have been developed by Federal agencies (U.S. Department of Agriculture 1994; Regional Interagency Executive Committee 1995), the State of Washington (Washington Forest Practices Board 1995), and the State of Idaho (Idaho Department of Lands 1995) to link land management with scientific assessments. These procedures are also being used on private lands in the States of California, Oregon and Montana as well as in British Columbia, Canada.

Although many similarities exist between these procedures, varying objectives have led to different methodologies for assessing watersheds. The federal watershed analysis process characterizes the human, aquatic, and terrestrial conditions and is more appropriately described as ecosystem analysis at the watershed scale. The federal analysis is driven primarily by issues identified prior to the start of the analysis. A list of approaches is provided for the interdisciplinary team of resource specialists, but no specific methodology has to be followed. The analysis is not a decision-making process, but establishes the context for subsequent planning (Regional Interagency Executive Committee 1995).

The Washington state and Idaho processes focus primarily on aquatic resources and the physical processes that influence aquatic habitat, but the Washington state process has more rigorous scientific analysis. Both processes have tightly choreographed methodologies that the analysts must follow. The Washington state process, however, requires scientific assessment of watershed processes by an interdisciplinary team of certified specialists. The Idaho process is designed to be implemented by a single trained resource manager. The analysis focuses on inherent hazards within a watershed and current stream conditions, rather than characterizing processes. Both procedures directly influence decision making through development of management prescriptions based on the watershed assessment.

Many people have looked to watershed analysis as a means for guiding and implementing watershed restoration, despite the fact that most of these processes are not explicitly developed to meet this objective. For watershed analysis to be useful in development of restoration plans, data must be gathered at a scale that can be utilized at the project level. In particular, any in-stream or riparian restoration work requires an understanding of watershed processes such as flooding history, sediment supply, and geomorphic context at a local scale. Of the three procedures described previously, the Washington state process is probably the best suited for prioritizing and implementing land management, monitoring and restoration projects because of its more rigorous use of scientific analysis and ability to affect management/restoration decisions. This paper outlines the general methods for conducting Washington state watershed analysis, describes restoration approaches, and examines how information from watershed analysis can be used to prioritize and implement restoration projects.

Washington State Procedure for Watershed Analysis

Watershed analysis is a regulatory process administered by the Washington State Department of Natural Resources (DNR) on state and private lands in Washington. The analysis is designed to address the cumulative effects of forest practices on the public resources of fish, water, and capital improvements (e.g., bridges and county roads). A watershed analysis can be initiated by either the DNR or voluntarily by a private landowner who owns more than 10% of a watershed. Watersheds range in size from approximately 100 to 200 km^2.

The Washington state watershed analysis procedure consists of four distinct components:

1. Resource assessment: Scientists identify hillslope hazards by assessing mass wasting, surface erosion, hydrology, and riparian condition. They also identify vulnerable resources by assessing fish habitat, stream channels, water quality, and capital improvements (Table 1). Sensitive areas of the watershed are delineated where hillslope hazards can affect a vulnerable resource (e.g., an area prone to landslides that delivers sediment to a fish-bearing stream).
2. Prescriptions: Land managers and scientists design prescriptions for each sensitive area. Prescriptions are methods for operating in sensitive areas to reduce or eliminate potential problems. Standard forest practice rules are applied on the remainder of the watershed.
3. Public review: The public is given the opportunity to review and comment on the findings through the State Environmental Policy Act (SEPA).

Table 1. Summary of watershed processes and resources addressed by the Washington state watershed analysis modules

Watershed analysis module	Watershed processes and resources addressed
Mass wasting	Debris torrents Landslides Earthflows
Surface erosion	Hillslope surface erosion - Gullying - Dry ravel - Sheetwash Road erosion
Hydrology	Peak streamflows
Riparian function	Large woody debris recruitment Shade / water temperature Bank stability
Channel condition	Historic channel disturbance Current channel condition Spatial distribution of channel response types Dominant habitat forming / geomorphic processes
Fish habitat	Distribution and relative abundance of salmonid fish Existing habitat condition Fish habitat utilization and preferences
Water supply / public works	Location and sensitivity of water supplies / public works - Public state roads and bridges - Reservoir, irrigation structures - Municipal, domestic, hatchery water supplies

4. Monitoring: A monitoring plan can be developed to track changes in watershed conditions and test the effectiveness of prescriptions. Monitoring is voluntary, although most landowners have initiated some monitoring following completion of the analysis.

Watershed analysis was developed by scientists and managers from state agencies, Indian tribes, and the timber industry working cooperatively under the state Timber Fish Wildlife agreement. Watershed analysis creates additional forest practice rules tailored to specific watersheds using a science-based assessment procedure. The assessment is typically conducted in 2 to 3 months by an interdisciplinary team of natural resource specialists that are certified by the DNR. The team may include geomorphologists, hydrologists, soil scientists, biologists, and other specialists as needed. The assessment provides information on physical and biological processes within a watershed such as the spatial distribution of processes, recent changes in the condition of the watershed, and how these changes influence aquatic resources.

The resource assessment information is passed onto a prescriptions team that is typically composed of land managers and some of the specialists that participated in the resource assessment phase. The prescription team develops various options for operating in sensitive areas to provide as much flexibility for landowners as possible while still protecting public resources. The assessment report and prescriptions must be approved by the DNR and go through public review. The entire watershed analysis process typically takes a full year to complete.

Monitoring is a critical component of watershed analysis because there are often limitations in the scientific assessment and many of the prescriptions are experimental in nature. Guidelines for developing

monitoring plans are provided in the watershed analysis manual (Washington Forest Practices Board 1995); however, monitoring is currently not required of landowners. A review of the watershed analyses, though, is required every 5 years and should provide information on the effectiveness of prescriptions in protecting resources. Analyses may also be reviewed sooner if fish habitat degrades, prescriptions are not working, or new scientific information is developed.

Washington state watershed analysis is designed to create options for conducting timber management activities in a manner that maintains natural rates of sediment input from hillslopes, minimizes potential changes in streamflow, and provides adequate riparian corridors for maintaining temperature, large woody debris, and nutrient levels. The following sections describe in more detail the procedures used to evaluate these watershed processes and relating the processes to regulatory prescriptions. For a complete listing of procedures, refer to the Washington state watershed analysis manual (Washington Forest Practices Board 1995).

Sediment

Mass wasting is the dominant source of sediment in many Pacific Northwest watersheds (Swanston and Dyrness 1973; Sidle et al. 1985; Megahan 1983). While landslides often initiate far above fish-bearing waters, debris flows, dam-break floods, and fluvial processes can route sediment downstream and affect fish habitat far below the landslide (Swanston et al. 1987). Both the volume and rate of sediment influx is considered in light of its effects on fish habitat.

Delivery of coarse sediment to streams is addressed primarily by the mass wasting module. The mass wasting analyst maps all historical mass wasting within the watershed as defined by the aerial photograph record and evaluates the sediment delivery potential to streams. A landslide inventory is produced and correlations are made with a number of variables including geology, slope gradient, slope form, and forest practices. The analyst uses these data to produce a mass wasting hazard map that identifies areas with the potential for mass wasting.

Another significant concern is fine sediment (<2 mm) from mass wasting, hillslope erosion, and road erosion that can reduce the viability of eggs in spawning gravel, reduce rearing habitat by filling the interstitial spaces of cobbles and gravel, and increase turbidity and nutrient levels in streams (Lisle and Hilton 1992; Lisle 1989; Chapman 1988). Fine sediment input into streams is addressed by the surface erosion and mass wasting modules. The surface erosion analyst evaluates recent harvest units and road construction in the field to examine the potential for gullying and sheetwash erosion. The road network is also evaluated using an empirically based model to estimate sediment production from roads. Predicted sediment yields are based primarily on road characteristics including traffic levels and road surfacing. Management-derived sediment inputs are compared to estimates of natural sediment input rates as part of a crude sediment budget to evaluate hazards. In this context, estimates of sediment input are useful for estimating *relative* contributions from the various sources of sediment in the watershed. Other sources of fine sediment from land use activities such as grazing and agriculture as well as episodic natural events such as fire are considered in the context of inputs from timber management activities.

Hydrology

Removal of forest cover can increase snow accumulation, allow for greater wind speeds, and increase solar radiation, thereby increasing the amount of snowmelt, especially during rain-on-snow events (Coffin and Harr 1992). Increasing the size and frequency of flood flows from timber harvest is a concern because larger and more frequent flood flows can change channel morphology through increased sediment and large woody debris transport and greater bank erosion. Larger and more frequent flood flows can also increase the depth of gravel bed scour, potentially destroying fish redds.

The hydrology module primarily addresses changes in peak flows during rain-on-snow events. The analysis procedure for peak flows consists of evaluating canopy coverage in relation to elevation. A modified U.S. Army Corps of Engineers (1956) empirical snowmelt model is used to estimate the increase in water available for runoff during various magnitude storm events. The water available for runoff is then related to streamflow using either gage data, field estimates of flood flows, or regional regression equations.

Riparian Corridors

Large Woody Debris

Large woody debris (LWD) is vital for maintaining fish habitats in most Cascade streams. Large woody debris dissipates stream energy, influences sediment storage and transport, and provides habitat and cover for fish both directly and through changes in channel morphology (i.e., pool formation)

(Fetherston et al. 1995; McMahon and Hartman 1989; Megahan 1982; Montgomery and Buffington 1993).

The riparian module addresses LWD recruitment to streams primarily by assessing the condition of riparian areas within 20 to 30 m on either side of streams. Forest stand type (coniferous, mixed, or deciduous), relative age, and density are measured for each stream reach from aerial photographs. The assessment concentrates on fish-bearing streams, but non-fish-bearing streams less than 20% gradient are also considered.

The potential for future LWD recruitment is considered together with the amount of present in-channel LWD to address concerns about present and future timber management. In-channel LWD data is collected in the field jointly with the channel and fish module team members. The riparian analyst interacts with the channel analyst to assess channel recruitment mechanisms such as meandering channels, debris flows, and bank erosion. The width of riparian recruitment can, thus, be extended beyond 20 m depending on site potential tree height and channel processes.

Temperature

Salmonids require relatively cool stream temperatures for all life history stages. During the summer when stream temperatures increase, cool temperatures are especially important for rearing juveniles and spawning adults. Empirical evidence in Washington shows that there is a strong relationship between canopy cover, elevation, and stream temperature (Sullivan et al. 1990). This has led to the development of a temperature screen that specifies the amount of canopy coverage needed at a given elevation to meet or exceed state water quality standards for stream temperature.

The riparian module analyst uses aerial photographs to assess canopy cover and topographic maps to establish elevations. All fish-bearing streams as well as perennial non-fish-bearing streams that contribute at least 20% of the flow to a fish-bearing stream are considered (Caldwell et al. 1991). At given elevation zones, targets for canopy coverage have been established through the state forest practice rules.

If canopy cover meets specified target levels, the stream is assumed to be below state maximum water temperature criteria at that location. If canopy cover does not meet specified target levels, it is assumed that maximum water temperature standards are exceeded. Available temperature data can be used to identify or verify problem areas.

Channel Condition

The channel condition module assesses past changes in channel morphology and processes, current channel conditions, and the potential sensitivity of channels to changes in inputs of sediment, wood, and water. Channel morphology reflects and integrates processes operating in a watershed because material eroded from hillslopes ultimately is delivered to and routed through the channel network. Channel and fish habitat condition, therefore, reflect the relative input of sediment, wood, and water relative to the ability of the channel to either transport or store these inputs.

The stream channel network is stratified into geomorphic units. Geomorphic units are reaches of streams that respond in a similar fashion based on comparable channel-forming processes. Alluvial fans or steep-gradient canyon tributaries would be examples of geomorphic units. These channel-forming processes are derived from the general geology and climate, which dictate stream gradient and confinement and hillslope topography and vegetation. Geomorphic units allow assessment of channel conditions on a watershed basis and provide a context for evaluating the influence of land management activities. These units become the basis for linking hillslope and channel processes during the synthesis portion of watershed analysis.

Fish Habitat

It is difficult to assess all the factors that affect salmonid production because of their wide-ranging life histories. Anadromous fish populations can vary substantially simply based on factors outside of their freshwater life history phase. For this reason, the fish habitat module assumes that evaluation of physical stream habitat characteristics will provide an adequate measurement of salmonid production during their freshwater life history phase. It is assumed that degradation of physical habitat features will result in reductions in salmonid production.

Two basic premises of the fish habitat evaluation are that: (1) physical habitat characteristics are strongly influenced by geomorphic setting, and (2) old-growth conditions most closely represent the conditions to which multiple species have adapted over the past several thousand years. This approach does not imply that preferred fish habitat only occurs in old-growth forests, but that knowledge of habitat in old-growth forests can form the basis for identifying changes in habitat conditions (Peterson et al. 1992). Indices of habitat conditions are based on

habitat utilization and on stream characteristics that have supported a multitude of species prior to human-induced habitat changes. Physical habitat features that are evaluated include depth and velocity ranges (grouped as channel units such as pools and riffles), pool frequency, pool size, cover, spawning gravel, and temperature ranges. Other elements of fisheries evaluated by the module include historic and current salmonid fish distribution, relative abundance of salmonids, and an assessment of factors limiting fish production.

Synthesis of Channel Condition and Fish Habitat

The channel condition and fish habitat module analysts jointly determine the vulnerability of fish habitat to changes in physical processes based on local channel conditions and fish biological requirements. The geomorphic units form the basis for assessing vulnerability because channel forming processes are assumed to be similar within a geomorphic unit. Vulnerability is assessed for the following processes: (1) debris torrents (i.e., debris flow and dam-break floods); (2) increases in coarse sediment; (3) increases in fine sediment; (4) changes in hydrology (primarily peak flows); (5) decreases in LWD and shade; and (6) removal of near-bank riparian vegetation.

The vulnerability of fish habitat is rated high, medium, or low, based on the channel sensitivity to a change in these processes and the potential impact on fish production. A channel geomorphic unit may not be particularly sensitive to a given input process, but if there is concentrated fish use (e.g., a chum salmon [*Oncorhynchus keta*] spawning reach) or limited habitat availability within that unit (e.g., a single area that accounts for most of the coho salmon [*Oncorhynchus kisutch*] winter rearing habitat), the vulnerability rating is changed to reflect the unit's importance for fish production.

Synthesis, Causal Mechanism Reports, and Prescriptions

An integral part of conducting watershed analysis is linking hillslope and channel processes. This may involve routing sediment derived from management-induced landslides through the channel network or discussing the role of wildfire in producing surface erosion and its consequent effect on fish production.

During the synthesis process, the resource assessment team identifies areas of resource sensitivity based on the likelihood of adverse change and deliverability to vulnerable resources. Some hazard areas identified by the resource assessment team may not be considered resource sensitive areas if impacts cannot be delivered to the resource of concern (e.g., unstable slopes that do not deliver sediment to streams). The resource sensitive areas are designated relative to the hazard area, rather than to the stream segments with the affected resource.

The resource assessment team rates both the likelihood of adverse change and the resource vulnerability as low, medium, or high. These ratings are placed in a management response matrix (Table 2) to provide direction on which situations need prescriptions and the standard to which prescriptions will be written.

There are three potential management responses (rule calls): (1) standard forest practice rules; (2) a prescription that minimizes potential impacts to the resource of concern; and (3) a prescription that prevents impacts that would damage or prevent the recovery of public resources. While the ratings end up simplifying complex technical problems, they provide an important link between scientific assessment and regulatory policy.

A primary tool for synthesizing the results of the resource assessment is the causal mechanism report. A causal mechanism report is produced for all areas of resource sensitivity. These reports provide a brief and focused summary of problem areas that can be used easily by the prescriptions team. The causal mechanism report summary has a brief problem statement, identifies linkages between land management and resources of concern, and provides additional comments to help guide the prescription team in developing appropriate options for timber management.

The prescription team develops management options consistent with the standard of protection required by the rule call in each causal mechanism report. The team must provide justifications for each prescription to ensure that the standards for protection are met. The DNR also provides technical review of the documents prior to initiation of the SEPA process to confirm that the watershed analysis meets regulatory standards.

Monitoring

Generally, three types of monitoring can be employed upon completion of watershed analysis: validation, trend, and effectiveness monitoring (MacDonald et al. 1991). Validation monitoring

Table 2. Management response for areas of resource sensitivity

Resource vulnerability	Likelihood of adverse change and deliverability		
	Low	Medium	High
Low	Standard rules	Standard rules	Prevent
Medium	Standard rules	Minimize	Prevent
High	Standard rules	Prevent	Prevent

assesses the scientific validity underlying elements of the resource assessment. Examples of validation monitoring might include testing of empirical models with local data or evaluating assumptions about woody debris input. Trend monitoring assesses resource conditions over time. Examples of trend monitoring evaluates could include collecting LWD data over time or measuring streamflows. Effectiveness monitoring evaluates whether prescriptions produced the desired result. Examples of effectiveness monitoring could include evaluation of prescriptions for road construction near unstable slopes or silvicultural prescriptions in riparian areas.

Monitoring may include basic research such as fish density estimates or stream gaging, gathering local data for the models used in the analysis, and/or monitoring the effectiveness of prescriptions. The latest version of the watershed analysis manual contains a monitoring module to assist in the development of a monitoring program. The monitoring program is typically developed in conjunction with all stakeholders in the watershed.

Watershed Analysis and Restoration

In addition to development of monitoring programs, data from the resource assessment can be valuable for developing restoration programs. The assessment results, however, are most useful if provided at a spatial and temporal scale that allows evaluation of the short- and long-term consequences of specific restoration actions. While watershed analysis in general has not been in use long enough to measure its utility for implementing restoration programs, the value of connecting the scale of scientific assessment to that of the restoration project can be illustrated by reviewing various approaches to watershed restoration. The following sections provide a brief overview of general approaches to restoration and an example of a specific framework for connecting the results of watershed analysis to watershed restoration.

Restoration Approaches

Restoration approaches can be divided into three general categories: (1) site specific, (2) conservation biology, and (3) landscape process. Site specific restoration approaches evaluate opportunities in a given location without explicitly considering the watershed context. Most restoration projects such as road abandonment, riparian plantings, and in-stream structure placement have simply used the site-specific approach. A given site might be considered deficient in certain elements and the objective of the restoration project is to provide or increase the productivity of those elements. The conservation biology approach tries to provide intact reserves (or patches) that maintain key habitat elements for a given species within a matrix of unsuitable habitat and has corridors to connect surrounding reserves (Forman and Godron 1986; Doppelt et al. 1993). The conservation biology approach has typically been applied to single species such as for northern spotted owl (*Strix occidentalis*) habitat conservation areas and bull trout (*Salvelinus confluentus*) recovery strategies, but could also be applied on a larger landscape level such as the national park system in the United States. The landscape process approach seeks to identify a natural disturbance regime for given elements in a landscape and provide enough of those habitat elements over the landscape to either mimic the disturbance regime or at a minimum provide a matrix of suitable habitat throughout the landscape. Very few examples exist for such a holistic restoration program, but data from watershed analysis would be particularly useful in identifying disturbance regimes for a landscape process approach.

The landscape process approach is not necessarily mutually exclusive of the other two approaches and may provide some significant advantages over previous approaches. Both the site-specific and conservation biology approaches could be important elements of an overall landscape process approach. Enough obvious site-specific problems exist that need immediate attention without the need for expensive, time-consuming landscape analysis. Similarly, a conservation biology approach can be an excellent short-term solution for protecting endangered species or maintaining important habitat elements of an ecosystem. Without a broader landscape approach, however, these other two approaches will not provide long-term sustainability for species. Many species such as anadromous fish, bull trout, and large terrestrial predators have enormous ranges that make it impractical to provide reserves over their entire habitat. Additionally, the conservation biology approach encourages intensive management outside of reserve areas that may significantly alter ecosystem function in the habitat of these species and potentially in the reserve areas. Moreover, any catastrophic disturbance within the reserve area would have disastrous consequences for these species.

The landscape process approach is not without its dangers, but it does provide a framework that is adaptable and potentially easier to implement. One of the greatest uncertainties with the landscape approach is our limited understanding of ecosystem processes. Many disturbance processes such as fire and floods occur infrequently and are difficult to evaluate over long time frames. In addition, the complex interactions within an ecosystem are poorly understood and the full impacts of a given action may not be anticipated. A critical component for implementing a landscape process approach is monitoring the effectiveness of restoration and having long-term data on disturbance processes.

Connecting Watershed Analysis and Restoration

Regardless of the restoration approach, five important elements are necessary to prioritize and help implement restoration activities:

1. identification of resources of greatest concern and their conditions;
2. identification of threats to resources and to the potential projects;
3. evaluation of the linkages between the threats and the resources;
4. a clear statement of the objectives of the project; and
5. identification of the time frame for expected recovery of the resources.

While few restoration programs currently incorporate all of these elements, a thorough evaluation using such a systematic approach could greatly improve the success of restoration projects. This evaluation is also critical for determining the costs and benefits of particular projects so that limited budgets are used effectively. Watershed analysis can and should provide the data necessary to objectively evaluate each of these elements.

Figure 1 contains an example restoration action plan based on the level of information generated by a TFW-type of watershed analysis. The resource of concern is identified specifically by stream segment, critical habitat element, and triggering mechanism for degradation of the habitat. The identification and prioritization of critical resource concerns should be established by the entire interdisciplinary team, ideally representing all watershed stakeholders. The threats to the resource are also specifically identified along with the confidence in linking the threat to the resource damage. This evaluation requires very site-specific information from the watershed analysis, but is critical for prioritizing restoration actions as well as monitoring programs. Finally, the time frame for recovery along with the action plan clearly identifies the length of time for results from the restoration program. This element is important for establishing the specific objectives of the program and providing realistic expectations for recovery of the resource.

Conclusions

Watershed analysis that is scientifically rigorous and has direct links to management decision making can provide useful data for landscape management, monitoring, and restoration. The scale at which watershed analysis is conducted is critical because long-term management and restoration requires an analysis of landscape elements throughout the range of the species of concern. Most restoration efforts to date have provided at best short-term improvements because they rarely evaluate the appropriate spatial and temporal scales for long-term recovery of watershed resources. Watershed analysis that provides data on watershed processes at a local scale, but within a larger spatial and longer temporal context can ensure that restoration efforts will be successful in the long term.

Resource of concern	Spring Chinook spawning habitat in Segments 12 and 13 of the Yakima River is degraded by fine sediment, lack of holding pools, and potential scour during high flows.
Threats to resource	1. Landslides from road built prior to 1980 directly into the Yakima River and from debris flows in Cabin Creek. 2. Surface erosion from extensive road network in Cabin Creek. 3. Reduction in large woody debris from power lines in riparian area. 4. Large proportion of watershed is hydrologically immature.
Confidence in linkage between threat and resource	1. High 2. High 3. Moderate 4. Low
Time frame for recovery	1. 50–100 years for sediment to route through system. 2. Indefinite, continual input of fines as roads are used. 3. Indefinite for large woody debris input unless power lines relocated. 4. 25 years for hydrologic maturity.
Action plan	1. Roads 2520 and 2550 will be abandoned with culverts removed and sidecast material pulled back. 2. Road 2500 will have any sidecast on slopes >60% pulled back. 3. Drainage problems and sediment delivery to streams will be evaluated and addressed for the entire road network. 4. Alternative silvicultural methods for maintaining vegetation under power lines will be investigated with the power company. 5. Placement of large woody debris to increase holding pool capacity will be explored.
Objective with time frame	The action plan will reduce sediment delivery to Segments 12 and 13, but measurable improvement in percentages of fines in gravels is not expected for 10 years or more. Short- and long-term (~50 years) large woody debris input will increase with silvicultural treatments and placement of wood in-channel.

Figure 1. Example restoration action plan.

Active restoration management will be necessary to speed the recovery of many aquatic systems, but will need to rely on good local data collected in the context of landscape processes. Due to the long-term nature of most recovery efforts, a systematic approach to restoration is also critical. Finally, monitoring of restoration projects and the recovery of resources will provide a critical feedback mechanism for future restoration efforts.

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Forest Ecosystem Management and Public Involvement: A Case Study in West-Central Alberta

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Abstract

The Grande Prairie Division of Canadian Forest Products Ltd. (Canfor) has formed a Forest Management Advisory Committee comprising representatives of community stakeholders, which includes local government, sporting groups, First Nations, the oil and gas industry, and Alberta Environmental Protection to participate in the development of its Forest Management Plan. Canfor has also formed a Forest Ecosystem Management Task Force comprising experts in the fields of ecology, forestry, and wildlife biology employed by academia, resource management consultants, and the Alberta Provincial Government to assist in establishing principals of sustainable ecosystem management that will be used as the foundation of the Forest Management Plan. Concurrent to this, Canfor is in the process of identifying research priorities pertaining to forest growth and yield, reforestation practices, landscape ecology, harvesting practices, wildlife habitats, and fisheries management to enable to effective interaction with academic, industrial, and government-sponsored research groups to initiate projects that will contribute to an understanding of the forest ecosystem. All these initiatives coincide with Canfor renewing its Forest Management Agreement with the Province of Alberta and Canfor's development of a 20-year Forest Management Plan for the Crown lands included in its Forest Management Area. Canfor anticipates the development of a Forest Management Plan that will have the flexibility to incorporate advances in our understanding of the forest ecosystem.

Gilmore, D.W. 1998. Forest ecosystem management and public involvement: a case study in west-central Alberta. Pages 55–65 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

The Government of Canada, with support from provincial and territorial governments, ratified the United Nations Convention on Biological Diversity in 1992, which led to the development of a Canadian Biodiversity Strategy (Biodiversity Convention Office 1995). The development of a Canadian Biodiversity Strategy is well-timed because it has the potential to serve as a focal point for provincial conservation strategies being developed and implemented across Canada (see Natural Resources Canada 1995). In essence, this strategy recognizes the global necessity of conserving biodiversity and using our biological resources wisely. A Forest Conservation Strategy for the Province of Alberta is in the final stages of development with the goal *"To maintain and enhance, for the long term, the extent and health of forest ecosystems in Alberta for the sake of all living things locally, provincially, nationally and globally, while providing environmental, economic, social and cultural benefits for present and future generations"* (Alberta Forest Conservation Strategy Steering Committee 1996). Canadian Forest Products Ltd. (Canfor) supports these federal and provincial strategies for maintaining viable and healthy forest ecosystems from the corporate (Nielssen 1995) to the division (Canfor, Grande Prairie Division Business Plan 1996, unpublished) level.

The development of these federal and provincial conservation strategies is unprecedented, and their implementation will undoubtedly affect the way forestry is practiced in Canada. The purpose of this paper is to present "a work in progress". Specifically, I outline the approach being used by the Grande Prairie Woodlands Division of Canfor to incorporate Forest Ecosystem Management (FEM) into the Forest Management Plan (FMP) being developed for the 655 485-ha Forest Management Area (FMA) under its tenure in west-central Alberta.

A Brief Primer on Canfor

Canadian Forest Products Ltd. is an integrated Canadian forest products company with its headquarters in Vancouver, British Columbia (B.C.). Canfor was founded in 1938 in New Westminister, B.C. as a furniture veneer plant employing 28 people. Canfor and affiliated companies currently employ approximately 5700 people. Canfor has extensive forest operations, which include 12 woodlands divisions, and manufacturing facilities on the south coast of B.C., including northern Vancouver Island, the northern interior of B.C., as well as in west-central and northwestern Alberta.

The Grande Prairie Woodlands Division of Canfor has a 20-yr FMA with the Province of Alberta that will be subject to renewal in 1998. In this agreement, Canfor was granted the coniferous timber rights for a 655 485-ha FMA. Fifty-one percent of the land base under Canfor's tenure is considered productive coniferous [spruce (*Picea* spp.), lodgepole pine (*Pinus contorta*), balsam fir (*Abies balsamea*)] forest. In 1995, Canfor harvested 654 556 m^3 of spruce–fir–pine from 2454 ha. Annual harvest levels are generally less than the government-approved annual allowable cut of 730 000 m^3. Approximately 173 million board feet of spruce/pine/fir lumber with pulpwood or chips as by-products are produced annually. A dual forest land tenure is shared with Tolko Industries Ltd. (Tolko), a B.C.-based integrated forest products company, on a 70 000-ha portion of the FMA. Tolko holds a Deciduous Timber Allocation to provide aspen (*Populus* spp.) fiber to its oriented strand board plant in High Prairie, Alberta. Oil and gas companies comprise the remainder of the industrial stakeholders within the FMA. Non-industrial stakeholders include outdoor sports, hunting, fishing, trapping, and recreational enthusiasts.

Supported Research Programs

Canfor supports several provincial and regional research initiatives, six of which are described here. Canfor is an active participant in the West-central Alberta Caribou Standing Committee. The purpose of this committee is to propose a habitat/supply methodology, operating guidelines for the forestry and oil and gas sector, and to outline a research agenda for a geographic region in west-central Alberta where three caribou (*Rangifer tarandus*) herds are located.

The Huallen Seed Orchard Company was formed by Canfor and four other industrial partners (Weyerhaeuser Canada Ltd., Alberta Newsprint Company, Millar Western Industries Ltd., and Weldwood of Canada Ltd.) and Alberta Environmental Protection (AEP), Land and Forest Service Division (LFS). The purpose of this cooperative is to produce genetically improved white spruce and lodgepole pine seed for inclusion in the west-central Alberta reforestation programs of the participating companies.

The Western Boreal Growth and Yield Cooperative is based out of the University of Alberta and is supported by forest product companies throughout Alberta by the LFS, as well as by a growth and yield cooperative based in B.C., and several companies from Saskatchewan, the Yukon Territories, and B.C. The current focus of this cooperative is growth and yield

research in mixed-wood (conifer and deciduous) stands. Canfor is also a member of a B.C.-based growth and yield co-operative. The Northern Industrial Vegetation Management Association has strong representation throughout B.C., with Canfor being one of two members from western Alberta. The purpose of this cooperative is to support applied silvicultural research with a focus on vegetation management.

Canfor is an active participant with AEP, Fish and Wildlife Division in a joint Co-Operative Fisheries Inventory. The purpose of this program is to collect base-line data on the status of fish populations within Canfor's FMA. This project is in its third and final year with a final report anticipated in late 1996 or early 1997.

Structure of the Forest Management Plan Development Process

Traditionally, public input has been solicited during the final development stages of a FMP (Beckley and Korber 1995). Public participation is key to the development of a successful ecologically based FMP. Recognizing this, Canfor actively solicited public participation in August of 1995 through the formation of a Forest Management Advisory Committee (Advisory Committee). This committee is composed of representatives from stakeholder organizations from the geographic region within which Canfor's Forest Management Area is located (Table 1). The Terms of Reference for the Advisory Committee are

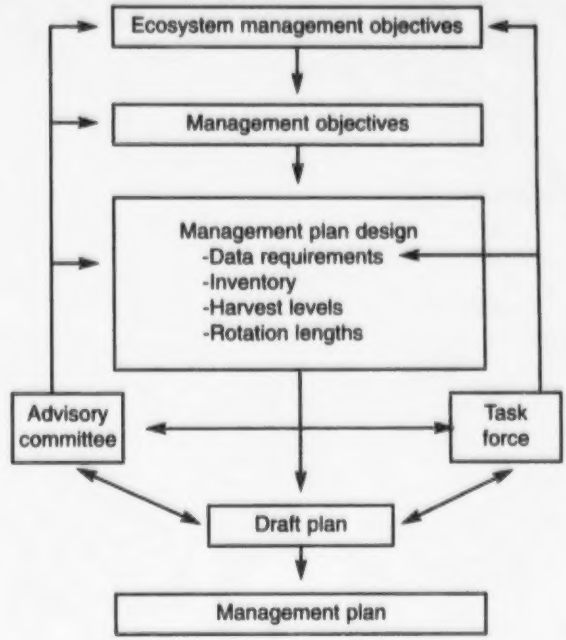

Figure 1. Schematic diagram of the relationship between the Forest Management Advisory Committee, the Forest Ecosystem Management Task Force, and the development of the Forest Management Plan.

Table 1. Stakeholder groups represented on the Forest Management Advisory Committee

Committee members
Alberta Logging Association
Canadian Association of Petroleum Producers
County of Grande Prairie
First Nations, Sturgeon Lake Band
Grande Prairie Chamber of Commerce
Grande Prairie Regional College
Metis Nation of Alberta, Local 1990
Municipal District of Greenview
Town of Valleyview
Valleyview Fish and Game Association
Advisory members
Alberta Environmental Protection, Land and Forest Service
Canadian Forest Products Ltd.
Tolko Industries Ltd.

Table 2. Members of the Forest Ecosystem Management Task Force

Mr. Harry Archibald Forest Ecologist	Alberta Environmental Protection Edmonton, AB
Dr. James A. Beck Habitat Supply Analyst	University of Alberta Edmonton, AB
Mr. Peter Blake Silviculturist	Canfor Grande Prairie, AB
Dr. W. Richard Dempster Resource Consultant	Simons Reid Collins Vancouver, BC
Dr. Daniel Farr Wildlife Biologist	Foothills Model Forest Hinton, AB
Dr. Daniel W. Gilmore Forest Ecologist	Canfor Grande Prairie, AB
Mr. David Heurveux Wildlife Biologist	Alberta Environmental Protection Grande Prairie, AB
Dr. Winifred B. Kessler Forest Ecologist	University of Northern British Columbia, Prince George, BC
Mr. James Maitland Chief Ranger	Alberta Environmental Protection Valleyview, AB
Dr. David H. McNabb Soil Physicist	Alberta Environmental Centre Vegreville, AB
Mr. Dennis Quintillio Director	Alberta Environmental Protection Edmonton, AB
Dr. Joan Snyder Forest Ecologist	Grande Prairie Regional College Grande Prairie, AB
Mr. Dwight Weeks Forest Planner	Canfor Grande Prairie, AB

to participate in the development of the FMP and FEM Objectives, and in partnership with Canfor, to refine and implement the public involvement program.

Canfor has also sought the advice of experts in the fields of ecology, forest management, and wildlife biology from government and academia during the early development stage of its FMP through the formation of a FEM Task Force (Task Force) (Table 2). The role of the FEM Task Force is to serve as a scientific technical group to Canfor in the development of the 1998 FMP. The Task Force will provide guidance to Canfor to help ensure that the FMP reflects a sound and practical approach to FEM.

Both the Task Force and Advisory Committee are providing input into the FMP design and will review the draft plan prior to its formal submission to the government (Fig. 1). Canfor and the government are both represented on the Task Force, and along with Tolko serve as an information resource to the Advisory Committee.

Current Limitations to Forest Ecosystem Management

In their present form, timber harvesting regulations that are designed to extract the most volume are not conducive to FEM. Historically, the allocation of the forest resource has been driven by the wood fiber requirements of industry. Consequently, FMPs on Crown lands in Alberta have been written to follow operating ground rules designed to extract the most timber volume. In brief, operating ground rules call for the harvest of older, high-risk (in danger from insect and disease, flooding, fire) stands first. All merchantable volume for the commercial species harvested is required to be removed from a cut block. Harvesting is usually scheduled under a two-pass system whereby 50% of the merchantable volume is removed from an operating area until the free to grow standard is achieved, after which the remaining volume can be harvested. Respective free to grow standards for lodgepole pine and white spruce require that they attain a 2-m and 1.5-m height, respectively, within 14 yrs.

During the 1990s, two works commissioned by the Government of Alberta have recommended changes to the present operating ground rules. The first, *A Report of the Expert Review Panel on Forest Management in Alberta* (Dancik et al. 1990) recommended that *"Government and FMA holders should use greater flexibility in determining which harvesting and regeneration system is best, after considering forest type, location, and land use priorities. Recognizing the need for land use zoning on the basis of management priorities, the system used should ensure that all land use interests are effectively integrated into forest management planning. This will require enlarging the body of expertise on forest planning and negotiating committees"* (Dancik et al. 1990, Recommendation 67).

General recommendations that can be reasonably extrapolated from the second report, a comprehensive biodiversity study in northeastern Alberta, include the maintenance of stand structure, the adoption of variable rotation ages, the adoption of a mixed-wood management model, and the establishment of mixed-wood forest reserves as ecological benchmarks (Stelfox 1995).

Two issues in regards to the present land tenure system between the Province of Alberta and FMA holders must be resolved before FEM can be fully implemented. Forest land tenure is granted for the duration of the FMA, which must be renewed every 20 yrs. This short-term land tenure arrangement (relative to the time required to grow a forest crop) precludes FMA holders from expending large amounts of capital on reforestation and stand tending (i.e., precommercial thinning) programs where monetary returns would not be realized until after their current land tenure agreement expired.

Table 3. Implications of ecosystem management

1. Ecosystem based
2. Ecologically sound human use (sustainability)
3. Mimic natural disturbance regimes
4. Viable populations of all native species
5. Large spatial scale
6. Long time horizon
7. Interagency coordination and public communication

Source: Galindo-Leal and Bunnell (1995).

Overlapping land tenure is another barrier to FEM. Canfor has the rights and responsibilities associated with managing the coniferous, but not the deciduous, component of the FMA under its current agreement with the Crown. There are provisions for conifer protection where Tolko has the rights and responsibilities associated with the harvesting of their deciduous allocation. In essence, this type of land tenure arrangement has resulted in two separate FMPs being prepared for the same forest landbase (one by Canfor for the coniferous volume and one by LFS for the deciduous volume). Management of pure coniferous and deciduous stands can be somewhat accommodated under this dual land tenure arrangement, but potential conflicts (both ecological and administrative) become apparent if one considers the management of mixed-species (coniferous and deciduous) stands under a dual land tenure system.

Foreseeable Challenges

Acceptance of the ecosystem approach establishes common ground for those concerned with forestry, wildlife, water, and recreation, thus encouraging partnerships in sustainability (Rowe 1992) and recognition that portions of the landscape will be allocated and managed for non-fiber producing attributes (Alberta Forest Conservation Steering Committee 1996). While there are numerous unknowns concerning the implementation of FEM (Atkinson 1995), seven broad implications are generally recognized (Table 3).

Canfor's concerns in regard to the implementation of FEM include: 1) public perception of ecosystem management, 2) our ability to quantify the landscape into ecological units at a spatially appropriate scale, 3) the effect of human-caused disturbances on

Table 4. Proposed Landscape Management Units (LMUs)

Kakawa Benchlands
Latornell–Lower Foothills
Latornell–Central Mixed-Wood
Iosegun Plains
Simonette Uplands
Simonette Benchlands
Deep Valley Plateau
Deep Valley Upland
Puskwaskau
Peace Upland
Peace Slopes
Riparian Zones (a special case LMU)

Note: LMUs are created from ecodistrict and ecosection delin eations based on biogeophysical similarities within Canfor's Forest Management Area. The names of the proposed LMUs listed above may change.

the Forest Management Area during the past in regards to natural disturbance regimes, 4) the viability of populations of endangered, threatened, or at risk species that occupy portions of the FMA, 5) integrating forest management and timber harvesting strategies with Tolko and other potential deciduous stakeholders, 6) the compatibility of oil and gas activity with FEM, and 7) the effect of FEM on the fiber supply to the Grande Prairie sawmill. I will describe the approaches being developed to address each of these concerns.

To begin, we must first clarify the concept of FEM. Numerous buzz-words or phrases have been developed and used during the past decade to describe FEM. New Forestry (Franklin 1989), New Perspectives (Kessler et al. 1992), Sustainable Forest Management (Boner 1995), Ecologically based Forestry (Kimmins 1995), Ecosystem Management (Rowe 1992; Grumbine 1994; Salwasser 1994; Galindo-Leal and Bunnell 1995), and the recently reintroduced term of conservation (Czech 1995) all advocate *a new emphasis in forest management, away from single-purpose timber production and toward a more holistic ecosystem orientation*. Stated another way, ecosystem management involves greater attention to what is retained in the stand or on the landscape, rather than what is removed (O'Hara et al. 1994). More (1996) maintained that FEM is a fuzzy concept, but recognized that an emphasis away from a precise definition of FEM can ultimately provide resource managers the flexibility necessary to implement FEM. Thomas and Dombeck (1996) and Thomas and Huke (1996) described FEM as, "an overall concept that influences the way we approach our work." FEM differs from the multiple-use philosophy of the 1970s in that all components of the ecosystem, including public participation are viewed as a collective management unit rather than as separate units (i.e., watersheds, viewsheds, fiber supply, wildlife habitat) to be managed simultaneously.

Defining what is ecologically good or acceptable is a genuine challenge. "We can only design 'ecologically sound management' or 'ecologically wise' activities based on prevailing social value systems" (Kimmins 1993). Ecology alone cannot be the basis for judgments as to whether or not a forest management activity, or natural disturbance is good or bad. Both the Advisory Committee and Task Force (Figure 1) are, and will be assisting us in presenting our rationale toward FEM to the general public.

The approach being used to divide the landscape into ecological units at a spatially appropriate scale has been designed to be compatible with the ecological land classification framework developed for Alberta. At a coarse scale, Alberta has been divided into natural regions and sub-regions based primarily on vegetation (Alberta Environmental Protection 1994). During the 1980s, two studies were undertaken in west-central Alberta to divide the landscape into

Table 5. Proposed criteria for evaluation within each Landscape Management Unit

Age-class distribution
Successional stage (includes old growth)
Natural disturbance types
Species composition
Riparian zones (inclusive of bogs and wetlands, and hydrology / watershed)
Landscape structure (interior habitat, patch size [i.e., stand size, pond size, muskeg size, etc.], habitat fragmentation [island remnants, habitat connectivity])
Soil-site productivity
Wildlife habitat (includes plants and animals) [target species, endangered species]

Note: Each criteria will be evaluated within each LMU, some criteria will not be applicable to every LMU.

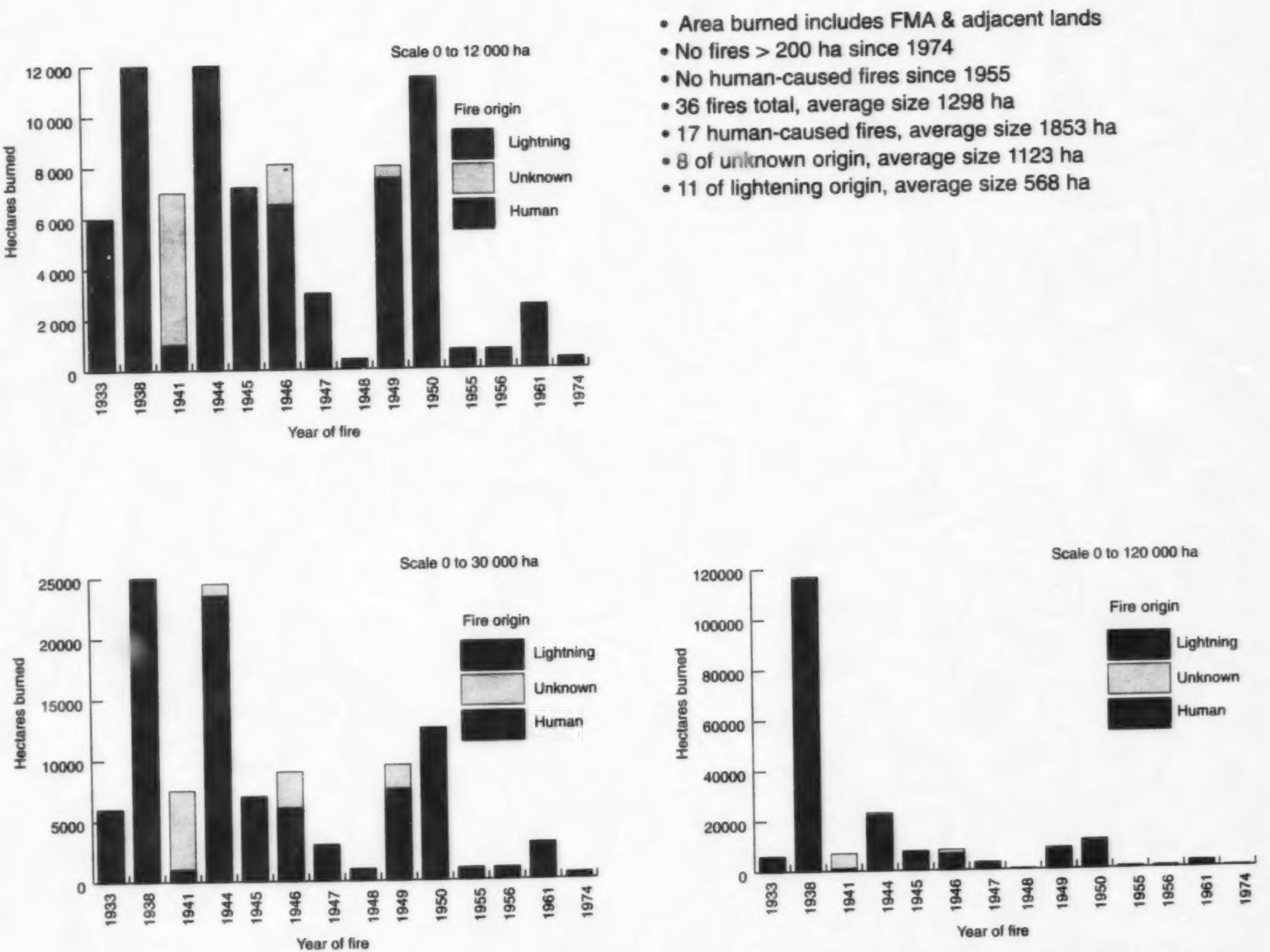

Figure 2. **Summary of wildfires (1931–1983) on Canfor's Forest Management Area greater than 200 ha in size (source: Delisle and Hall 1987).**

ecodistricts and ecosections (Nelson 1983; Archibald et al. 1984). The ecodistrict level of landscape delineation has been found to be the most useful to us for incorporating FEM into Canfor's FMP. At the ecodistrict level, this classification system has been refined to delineate mapping units based on topographic or edaphic features such as topographic position, elevation, slope, aspect, parent material, and soil drainage. Ecodistricts are unique in regards to their geographic location in Alberta (e.g., an ecodistrict found in the Lower Foothills Natural Subregion would not be found in the Prairie Parkland Natural Subregion).

There are twelve proposed ecodistricts within the FMA (Table 4). We anticipate combining one or more ecodistricts due to their similarities. Therefore, the term Landscape Management Unit (LMU) has been chosen to represent a single, or combined number of ecodistricts. The term LMU may be renamed, however, as the development of the FMP progresses. Addressing ecological and sociological issues by LMU allows Canfor to incorporate the collective knowledge of the Advisory Committee, the Task Force, and employees of Canfor in the design stage of the FMP.

Criteria consistent with those being developed by the Canadian Standards Association (CSA) standard for sustainable forest management (CSA 1996) are being developed for consideration within each LMU. Not all criteria will be applicable to each LMU. Canfor and the Task Force are in the process of defining, determining a process for measuring and monitoring, and addressing management issues associated with each criteria (Table 5).

Note that riparian zones are listed as both an LMU and a criteria (Tables 4 and 5). Canfor is using the definition of riparian zone provided by Dunster and Dunster (1996), *"those terrestrial areas where the vegetation complex and microclimate conditions are products of the combined presence and influence of perennial and/or intermittent water, associated high water tables, and soils that exhibit some wetness characteristics."* Canfor and the Task Force consider riparian zones to be a special case LMU in that large and obvious LMUs can be mapped, particularly along water courses. Smaller riparian zones that will be identified and site-specific management plans developed as part of the annual operating plan.

A portion of Canfor's FMA was not included in the mapping and classification programs of the 1980s due to access limitations at the time. Canfor has recently had the remaining portion of the FMA classified to the ecodistrict level (Strong 1996) to be compatible with prior works (Nelson 1983; Archibald et al. 1984).

Human-caused disturbances have impacted natural disturbance patterns in a variety of ways. One of the natural disturbances most noticeably influenced by humans is fire (Fig. 2). Between 1930 and 1950, the majority of fires in the area currently encompassed by Canfor's FMA were of human origin. Fire suppression and public awareness programs have successfully reduced the number and size of human-caused and total fires during the past 40 yrs (Stelfox 1995). A 30-yr legacy of timber harvesting (from Canfor and prior forest land tenure holders) has also left its mark on the landscape.

Canfor will identify forest management issues that may affect endangered, threatened, or at risk plant and animal species during the development of its management plan and during the development of annual operating plans. Habitat management focusing on individual species that are endangered or at risk is not the best tool to reverse population declines (Kessler 1994; Caza 1995). Therefore, timber harvesting activity near environmentally sensitive areas (i.e., salt licks, water source areas, riparian zones) will be tailored to minimize the risk of population declines for the plethora of species that are not endangered or threatened and use these habitats.

Canfor and Tolko have developed integrated annual operating plans for the past two timber harvesting seasons with the encouragement and support of AEP, LFS. Canfor is striving to integrate its long-term (20-yr) planning with both Tolko and the oil and gas sector, in part, through their participation on the Advisory Committee (Table 1).

Canfor will be implementing FEM within the framework of the Alberta Forest Conservation Strategy. This strategy incorporates the basic elements of the Triad Approach to Forest Land Allocation (Seymour and Hunter 1992). In brief, this approach to forest management involves the allocation of lands into 1) ecological reserves or benchmarks; 2) extensive, New Forestry, or FEM zones; and 3) high-yield, intensively managed tree farm zones.

As described earlier, 51% of the FMA is considered to be productive coniferous forest. It is also worth noting that the productive landbase from a fiber perspective will increase if stands dominated by deciduous species are included in the inventory. In this context, we are considering a combination of two types of ecological reserves, or benchmark selection strategies. The first type of ecological benchmark

being considered involves the selection of areas never to be harvested. These areas may encompass areas considered to be both nonproductive and productive from a fiber perspective. In the present FMP, about 6% of the productive coniferous forest land base was placed in this category to account for riparian zones that would not be harvested. The second type of ecological benchmark being considered involves allowing harvested areas to recover from timber harvesting with minimal, or no post-harvest intervention (i.e., site preparation, reforestation). Parallel to this, extended rotation ages (>60 yrs and up to 250 yrs) may be prescribed for some areas. This type of benchmarking would allow us to monitor ecological processes and conduct ecological comparisons between areas that were harvested and not reforested, harvested and reforested, and those not harvested.

Fiber production will be the primary focus in the high-yield, intensively managed tree farms zones, which will be selected based on their potential productivity. A full range of silvicultural tools will be applied to maximize yields under short (<60 yr) rotation periods. Intensive site preparation, genetically improved planting stock, post-planting vegetation management (both mechanical and chemical), and pre-commercial and commercial thinning will be among the tools available to the resource manager in the tree farm zone. Canfor is currently realizing about an 8% gain in potential fiber productivity through its tree improvement program in association with the Huallen Seed Orchard Company. It will be possible to realize gains in productivity from the advanced generation breeding program for lodgepole pine and white spruce of between 30 and 35% (Beck and Beck 1996). An additional 30% gain in productivity may be realized from precommerical thinnings (Peter Blake, Canfor, Grande Prairie, Alberta, personal communication). In summary, a commitment to intensive forest management on a portion of the landbase would result in a win-win situation whereby increased yields from the intensively managed areas would increase the management options for resource managers on the remainder of the land base (Seymour and McCormack 1989). Since FMAs are renewed every 20 yrs, the security of forest land tenure issue must be resolved prior to any long-term commitments from industry concerning intensive forest management.

Creative and innovative silvicultural systems implemented at the stand and landscape level will play an important role within the framework of FEM. Silvicultural options include the implementation of a range of even-aged (Seymour and Hunter 1992) and uneven-aged (Guldin 1996) systems that can be designed to produce an infinite variety of stand structures (O'Hara et al. 1994). Canfor in cooperation with LFS will be determining what type of forest management activities will be applied and what silvicultural tools will be applied in the New Forestry, FEM, or the extensive management zone as defined in the Alberta Forest Conservation Strategy. The relative percentages of land allocated to each leg of The Triad will be determined by Canfor in cooperation with LFS and with advice from the Advisory Committee and the Task Force.

Successful implementation of FEM offers several opportunities. The FEM will demonstrate Canfor's commitment to sound forest stewardship, which will likely be a factor in future land tenure renewals on Crown lands. The FEM will assist Canfor in producing products that will be certified as originating from a sustainable forest, which will become an increasingly import marketing tool (Upton and Bass 1996). Finally, and most importantly, the FEM will play a role in maintaining the integrity of the forest resources of Alberta, a benefit to be shared among all Canadians.

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Environmental Changes Affecting the Population Density of Brown Trout (*Salmo trutta*) in the River Isojoki Basin, Western Finland

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Abstract

Population densities of brown trout older than young-of-the-year were examined on the basis of habitat and catchment factors in the brooks of the River Isojoki basin, western Finland. Forestry is the main activity that has affected those environmental factors. A significant multivariate regression model was achieved, where single significant ($p<0.05$) independent variables were abundance of pools, abundance of undercut banks, pH value of water, and (negatively correlated) abundance of ditches in the catchment area. These four factors explained 42% of the variation. The population densities of brown trout in the dredged (channelized) brooks were compared with those of natural ones. The mean population densities in the dredged and natural sites were 4.7 and 11.0 trout/100 m^2, respectively. The transformed means differed significantly from each other ($p = 0.03$). The multivariate regression model explained 90% of the variation of trout density of natural sites. The determination coefficient of environmental variables in dredged sites was low, at 29%. The factors representing physical diversity of the brook channels had the most pronounced influence on the population density of brown trout in the forest brooks. A direct demonstration of this is the lower population density of trout in dredged sites as compared to natural ones. Acidity (pH value) influences the distribution of fish and is a limiting factor for their reproduction. The lowest pH values were less than 5. The negative influence of drainage on the population density of brown trout may be caused by permanent changes in water quality, by changes resulting from increased sediment load, or by changes in the composition of the bottom material in the brook channel. Hydrological changes may produce similar effects. None of these factors can be excluded on the basis of these results.

Ahvonen, A. and E. Jutila. 1998. Environmental changes affecting the population density of brown trout (*Salmo trutta*) in the River Isojoki basin, western Finland. [illegible]

Introduction

Despite the extensiveness of forestry in Finland, the number of studies conducted on the effects of forestry on fish and fisheries in running waters are limited. Some studies have been made on the effects of forest drainage on ascending behavior of salmonid fish, on smolt production, and on the filling of spawning grounds with sand (e.g., Viitala and Hyvärinen 1986; Kännö 1981). A few studies about the effects of forestry on fish and fisheries were done in Scandinavia as well (e.g., Bergquist et al. 1984; Simonsson 1987). Most studies on this subject have been done in North America. However, North American results do not necessarily apply to Finnish circumstances because climate, soil, geomorphology, tree species, harvesting practices, characteristics of watercourses, and fish stocks are different.

In this study, we examined the effects of some forestry activities on brown trout (*Salmo trutta L.*) densities in a basin where the proportion of forests and bogs in the total catchment area was high. The study was done in the River Isojoki basin, western Finland, which has a remarkable value both for fishery and for the conservation of the biodiversity of the brown trout stocks. The questions in the study were: 1) Could trout density of the brooks be explained by environmental factors? 2) What might these factors be? and 3) Can these factors be considered as forestry factors or as factors that were altered by forestry?

Study Area

The River Isojoki is located in the western Finland (62°N, 22°E), and it flows to the Gulf of Bothnia in the northern part of the Baltic Sea. The catchment area of the river is 1117 km^2. About one third of this area was included in this study. The main stream of the river is about 75 km long, and it has three main tributaries. One of the tributaries and its catchment (River Kärjenjoki area) was excluded, because it is known that no natural trout stocks exist there. A typical characteristic of the River Isojoki system is a large number of brooks (over 50), compared with other Finnish river systems of the same size.

The main land use of the study area (forests and bogs) was forestry at about 97%. The share of agricultural use was about 2%. Besides logging and harvesting, forestry in the study area includes forest improvement. The most pronounced improvement method during the last three to four decades has been forest drainage. Drainage is used to enhance tree growth by improving moisture conditions because the inclination gradients are low. In the study area a typical gradient is 3–4 m/1 km. The mean abundance of ditches was about 10 km/km^2. Dredging of the brooks has been a drainage method used in connection with ditch excavating. Dredging involves excavating the brooks to be deeper, wider, or straighter to carry water from the ditches more effectively. Dredgings were carried out during the 1960s and 1970s.

Material and Methods

The brooks of the River Isojoki were surveyed by electrofishing to estimate (Bohlin et al. 1989) brown trout density, the dependent variable in the analyses. The sample site locations were selected on the basis of maximum relative silvicultural and forestry land use of the upper catchment, and according to the existence of potential brown trout habitats, i.e., rapids. The number of sample sites in one brook varied from one to five. The minimum distance between sites were about 500 m. In the analysis there were 58 sampled sites representing 41 different brooks. Young-of-the-year parr were excluded from the analysis, because of low or sporadic catchability. Brown trout was, in practice, the only fish species obtained in the catches.

The sampled area, mean depth, mean surface velocity, conductivity, and pH value of the water were measured. The distribution of bottom material [sand, gravel, stones (<2 cm, 2–10 cm, 10–30 cm, >30 cm), and clay], distribution of surface velocity (<0.2 m/s, 0.2–0.4 m/s, 0.4–0.7 m/s, >0.7 m/s), coverage of bottom vegetation, and shading of trees in the site were estimated visually. Also, the abundance of bottom pools (minimum depth 30 cm, minimum diameter 50 cm), undercut banks (minimum depth 20 cm, minimum length 100 cm), and stream cascades (minimum fall 15 cm) were estimated visually. The sample sites were classified as either dredged or natural. There were in some cases both dredged and natural sample sites in the same brook.

A cartographic survey was made to measure the area of upper catchment for each sample site, the land use or land type (e.g., forest, bog, and agriculture) and total length of ditches in the catchment.

A multivariate regression model was used to analyze the variation in brown trout densities. The independent variables were first nominated on the basis of correlation analyses. After that, the potential independent variables were selected for the final regression models. To linearize the variables and to normalize the distributions, the variables were [illegible] or log(x)-transformed. The normality

independency, equality of variances, and autocorrelation of variables or residuals were controlled. Statistical analyses were calculated with SYSTAT (1992) programs.

Results

In a regression model for all fishing sites (Table 1), the brown trout density was determined with four significant variables ($p \leq 0.05$). The variables were the abundance of pools, abundance of undercut banks, pH value, and relative total length of ditches in the upper catchment. The last one was the only negatively regressioned variable. These four variables determined 42% of the variation in brown trout densities.

Trout density in natural sites (mean 11 trout/100 m^2) was clearly higher than in the dredged sites (4.7 trout/100 m^2). The log-transformed means differed significantly ($|t| = 2.166$, DF = 74, $p = 0.03$). Therefore, sample site results from natural and dredged sites were also analyzed separately.

Table 1. Regression model of brown trout density [log(x+1)] of all sample sites in streams. $N = 58$, $F = 9.795$, $p<0.01$, $R^2 = 0.42$, $D = 1.946$ = Durbin–Watson-value fir independency of residuals, $r = 0.001$ = autocorrelation.

Independent	Coefficient	Standard error	Standard coefficient	T	P
Constant	-1.371	1.076	0.000	-1.274	0.21
Pools	1.022	0.261	0.414	3.919	<0.01
pH value	0.444	0.157	0.308	2.834	0.01
Undercut banks	0.594	0.254	0.258	2.343	0.02
Upper ditches	-0.494	0.250	-0.217	-1.979	0.05

Table 2. Regression model of brown trout density [log(x+1)]from sample sites in natural streams. $N = 19$, $F = 22.47$, $p<0.001$, $R^2 = 0.90$, $D = 2.111$ = Durbin–Watson-valueor independency of residuals, $r = -0.144$ = autocorrelation.

Independent	Coefficient	Standard error	Standard coefficient	T	P
Constant	2.080	1.590	0.000	1.309	0.21
Pools	1.064	0.262	0.452	4.056	<0.01
Proportion of bog	-3.373	0.907	-0.337	-3.716	<0.01
Shading	-2.151	0.759	-0.307	-2.834	0.01
Mean depth	0.613	0.247	0.230	2.485	0.03
pH value	0.289	0.127	0.212	2.265	0.04

Table 3. Regression model of brown trout density [log(x+1)] from sample sites in dredged streams. $N = 40$, $F = 4.732$, $p = 0.01$, $R^2 = 0.28$, $D = 1.604$ = Durbin-Watson-value for independency of residuals, $r = 0.129$ = autocorrelation.

Independent	Coefficient	Standard error	Standard coefficient	T	P
Constant	-2.169	1.424	0.000	-1.522	0.14
Undercut banks	0.770	0.296	0.380	2.627	0.01
pH value	0.422	0.218	0.279	1.933	0.06
Pools	0.356	0.253	0.229	1.409	0.12

The regression model for natural sites was highly significant, and five independent variables could determine 90% of brown trout density variation (Table 2). Positively regressioned independent variables were the abundance of pools, mean depth of the fishing site, and pH value. Negatively regressioned variables were the proportion of bog in the upper catchment and shading of the site by trees.

The ability of independent variables to determine brown trout density in dredged sites was considerably lower than in the natural sites. The only single significant variable in the model of dredged sites was the abundance of undercut banks. In addition, pH value ($p = 0.06$) and abundance of pools ($p = 0.12$) were retained in the model because they could improve the coefficient of determination by 12% units for a total of 28% (Table 3).

In all the cases, other parameters (see Material and Methods) than those accepted in the models did not provide a significant determination in the variation of trout density.

Discussion

The factors representing physical diversity of the brook channel had the most pronounced influence on the population density of brown trout in the forest brooks. A direct demonstration of that was the lower population density of trout in the dredged sites as compared with the natural ones. In the regression models, the variation in trout density was most effectively explained by the abundance of pools and undercut banks. Those factors can be considered to describe well the morphometric diversity of the brook channel, especially with regard to the habitat needs of brown trout. The lower trout density in the dredged sites can also be considered a consequence of forestry, because brook dredging was formerly a common practice in connection to ditching, to improve forest soil drainage in Finland.

The presence of gravel is often associated with trout populations. In this study, however, gravel was not an explaining factor for trout density. The same applies to other variables describing bottom quality (e.g., substrates, bottom vegetation), and to variables of water velocity. These factors appeared to be more important for young-of-the-year parr. In stocking experiments made in these brooks, stony bottom material (2–10 cm in diameter) positively affected the population density of parr. Aquatic vegetation had similar effect on the parr biomass (Jutila et al. 1998).

In all models, pH value was a significant determinator of brown trout density. Acidity is a well-known limiting factor in fish species' density or existence (e.g., Lappalainen et al. 1988; Rees and Ribbens 1995). In the data, there were cases of very low pH values (to 4.3). No general limiting pH values can be expressed, because the effect of acidity depends on the other contemporary water chemical values (e.g., Vuorinen et al. 1993). On the other hand, local fish stocks can, to some extent, adapt to acid circumstances (e.g., Grande et al. 1978; Tuunainen et al. 1991).

Shading is often considered to be a positive habitat factor for salmonids. In this study, the shading was found to be negatively correlated with population densities of brown trout. To have a positive effect on trout habitats, the vegetation shading should occur just over the water surface, where it is often described as a cover (e.g., Heggenes 1988). In this study, the cause of shading was identified as streamside trees. Streamside clearcutting can decrease that shading, and decreased shading can increase water temperature significantly (Ahtiainen 1990). This phenomenon was observed also in the upper brooks of the River Isojoki, although in the sample sites, the lack of shading was in some cases natural and in some cases caused by timber harvesting. In the conditions of the upper brooks of Isojoki, the increased water temperature can have a positive influence on primary production and even on trout stocks because the water originates in many brooks from groundwater and is often relatively cold even in summertime.

In the model of all fished sites, one negatively regressioned determinator of the trout density was the total length of ditches in the upper catchment. Ditch excavating is a regular silvicultural activity in Finland, and in the study area the mean abundance of ditches was 10 km/km^2. The negative effect of the drainage can be explained by permanent changes in the water quality, by morphological changes, or bottom material changes caused by increased sediment loads from the ditches, and even by the hydrological changes caused by the ditches (Ahonen et al. 1992). None of these potential affecting mechanisms could be excluded on the basis of this study.

In general, the results showed that forestry has affected the population densities of brown trout. The negative effects of forestry were found to be long lasting, and the effects could be even considered permanent when radical methods, such as brook dredging, had been in use.

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The Upper Little Smoky: Integrating Timber Harvesting with Fisheries and Recreation Management – A Case Study

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Abstract

The upper Little Smoky River has long been considered a popular sport fishing and wilderness recreation area. Alberta Newsprint Company (ANC) is proposing an integrated approach to allow timber harvesting to occur without compromising the integrity of the fisheries or the capacity of the area to provide an outdoor recreation experience. The paper presented would provide an opportunity to review ANC's plan and will hopefully generate thought and discussion on the potential for integrating industrial activity, recreation use, and fishereis habitat. Alberta Newsprint Company's intention is the integrate stakeholder concerns into the plan as much as possible with the plan having a firm basis in scientific principles. It is expected that the plan will include a variety of management scenarios ranging from special logging techniques to special access management considerations. The plan will also include an assessment program to determine if these special practices are having the affect that was intended.

Introduction

As part of a commitment to sustainable forest management, Alberta Newsprint Company (ANC) is developing a management strategy for the Little Smoky River area. The Little Smoky's various resource values include timber, caribou, wildland recreation, sport fisheries, and oil and gas reserves. Many recent changes have led to a need for a comprehensive planning approach for key areas within Alberta. These changes include increased resource consumption, more pressure on the land base for non-consumptive type uses, increased desire by the public to be involved in land use decisions, and increased awareness of the needs of certain fish and wildlife species. Provincial government reorganization has also created changes in how land use decisions are made and by whom. The approach used by ANC is to integrate these diverse resource needs in order to maximize all stakeholders' benefits.

The Little Smoky area falls within ANC's Forest Management Agreement (FMA) area. The FMA is 381 000 ha in size and supports a net annual allowable cut (AAC) of 570 193 m^3 comprising 66.4% of the coniferous AAC and 100% of the deciduous. The balance of the coniferous fibre rights are held by three other forest products companies, namely Millar Western, West Fraser, and Mostowich Lumber.

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Little Smoky Area Description

Primary Vegetation and Landforms

The topography of the study area is characterized by broad rolling upland bisected by the Little Smoky River valley. A number of tributaries also have well-defined valleys. The western portion of the area is made up of more rugged foothills and a small subalpine area. Much of the area along the Little Smoky is flat muskeg. The Little Smoky area is divided into three ecological regions as defined by the Natural Regions Map of Alberta (Strong and Leggat 1992). Approximately 75 000 ha are upper foothills, 25 000 ha are lower foothills, and 2000 ha are subalpine. The lower foothills are dominated by closed canopied deciduous and coniferous mixedwood forest with aspen poplar, balsam poplar, white birch, white and black spruce, balsam fir, and lodgepole pine. The subalpine ecoregion is dominated by lodgepole pine with Engelmann spruce and subalpine fir as climax species.

Soils

The majority of the study area is covered by the Little Smoky Plateau, Berland Plateau, and Mayberne Plateau (Knapik and Lindsay 1983). Knapik and Lindsay (1983) describe these areas as being generally broken into three landscape types. There are fairly level plateau tops and benches, steep sloping escarpments, and valleys. The most common soils in the plateaus are Brunisolic and Podzolic Gray Luvisols developed on till. Where tills are discontinuous or absent, the soils are developed on gravel or soft bedrock. Well-drained Brunisolic and Podzolic Gray Luvisols on gravel and poorly drained peaty Gleysols on gravel are common. Organic soils on fens and bogs are also very common on the plateau areas. The steeply sloping escarpments are made up of Brunisols, Luvisols, and Regosols on colluvium and till. The valleys have soils developed on glaciofluvial and organic materials most commonly. Brunisolic Gray Luvisols on sands and Brunisols and Podzolic Gray Luvisols on gravel are also found. Poorly drained Gleysols are also common, as are large areas of organic soils on bogs. Regosolic and Gleysolic soils are found on active flood plains of creeks and rivers.

Climate

In general the climate is characterized by relatively short, cool summers and long, cold winters. Mean annual precipitation ranges from 50 to 750 mm, of which 60% falls as rain (Alberta Forestry, Lands and Wildlife 1988). Climate plays a major role in influencing the overall distribution of vegetation in the area. Winter generally extends from mid-November to late March, with frequent freeze–thaw periods ranging from 1 day to 2 weeks. Frost is a possibility during any month of the year.

Fisheries and Watershed

The primary drainage from the area is the Little Smoky River. Several large streams drain into the Little Smoky, as do many large muskeg areas. The most common fish species found in the Little Smoky area are Arctic grayling, rainbow trout, bull trout, and mountain whitefish (Bishop et al. 1979).

Wildlife

There are an estimated 250 wildlife species in the area. Key large mammal species include moose, elk, grizzly bear, black bear, white-tailed deer, and mule deer. A small herd of woodland caribou may be resident year-round in the study area. Woodland caribou are presently classified as endangered in Alberta because of concern that populations are in decline. Small furbearers include pine marten, fisher, wolverine, weasel, red squirrel, woodchuck, rabbit, lynx, coyote, wolf, mink, muskrat, beaver, and otter (Alberta Forestry, Lands and Wildlife 1988). Upland game birds include ruffed grouse, spruce grouse, and a few sharp-tailed grouse. Waterfowl populations are low, due to limited habitat (Alberta Forestry, Lands and Wildlife 1988).

Land Uses

Oil and Gas

Oil and gas exploration and development has been a moderate land use in the Little Smoky area. One major processing facility exists at the eastern edge. Seismic lines are prevalent throughout, while pipelines, well sites, transmission lines, and roads tend to be located in the more southerly portions. One major pipeline crosses through the center from northwest to southeast.

Trappers

Trapline dispositions cover the entire study area. There are seven traplines and trappers operating within the Little Smoky area.

Recreation

Extensive, dispersed outdoor recreation activities such as camping, fishing, hunting, snowmobiling, all-terrain vehicle use, and canoeing occur within the area. Recreation use tends to be associated with the Little Smoky River corridor itself. No formal recreation facilities exist, with the exception of several random camping sites that tend to be reused each year.

Timber Harvesting

Four forest product companies hold tenure in the study area. Alberta Newsprint Company Timber Ltd. holds FMA #8900026. Blue Ridge Lumber (1981) Ltd., Millar Western, and Mostowich Lumber hold Commercial Timber Quota numbers CTQ WO10004, CTQ WO10002, and CTQ WO1005, respectively. Harvesting to date by these four companies has been minimal within the study area. Historically, however, there has been considerable harvesting along the Little Smoky River.

Integrated Land-Use Strategy

A fundamental principle in the development of a land-use strategy for the Little Smoky River area is to use scientifically based information wherever and whenever possible. This recognizes that complete information may not be available in all cases, yet it attempts to distinguish between personal opinion and accepted scientific principles as the basis for decisions. Another fundamental principle is that of an adaptive approach. As new, scientifically based information becomes available, changes will be made to incorporate the information into the decision process.

As mentioned, one of the driving forces behind this approach to developing a land-use strategy is the increased desire of many public groups to be involved in making land-use decisions. Public and stakeholder concerns will be solicited and integrated into the process as much as possible. This is seen as an ongoing process, relying on both informal input opportunities and formal sessions with identifiable key interest groups. To date a series of presentation sessions have been held with the Fox Creek town council, Municipal District #16 council, and the chair of the Little Smoky River Boreal Forest Primitive Area Committee. Further meetings are scheduled with other stakeholder groups to attempt to gauge their respective interests in the Little Smoky region. A stakeholder general meeting has also been scheduled to gather all stakeholders together to have a collective discussion of concerns and potential planning strategies. Alberta Newsprint Company, as the FMA holder, will be responsible for the strategy development.

The public interests are also considered through a detailed approval process for all land-use activities on publicly owned land within the province of Alberta. The details of that approval process are beyond the scope of this paper and involve provincial and federal government departments, including Alberta Environmental Protection, Alberta Environment, Occupational Health and Safety, and the Marine division of Transport Canada.

Key Land Uses

Several key resource issues must be addressed within the context of a land-use strategy for the Little Smoky River area.

Caribou

A small herd of approximately 80 woodland caribou are resident year-round in the Little Smoky area. Caribou populations in this area are thought to have declined in the past 80 years; hence this species has been classified as endangered. Reasons for caribou decline are complex and difficult to assess, but predation (both human and nonhuman), access development, linear disturbance, and habitat alteration are all thought to affect caribou. The caribou habitat supply issue will be primarily dealt with through involvement in and direction from the West Central Caribou Standing Committee. To date, ANC has concentrated their information-gathering efforts on caribou habitat, given the potential alteration that timber harvesting will produce. The vegetation within the Little Smoky area has been described in detail and supplemented with a caribou habitat assessment. This assessment involved the placement of 226 plots across the study area to determine the quality and quantity of both arboreal and terrestrial lichens, which are significant components of the woodland caribou diet. Results from the inventory are being used to develop habitat suitability indices for use in predictive modeling to assist in decisions about land use.

A caribou habitat management approach is feasible but will require time and adaptability to consolidate and integrate with other land-management practices.

Backcountry Recreation

Fundamental to the development of this strategy will be an assessment of backcountry wildland recreation opportunities. These will include: view scape assessments, unique geologic and historical features assessments, potential campsite locations, and human travel corridors, including roads, trails, and waterways. The public involvement component of the strategy development will be critical in capturing the wants and desires of those wishing to utilize the recreational opportunities available in the area.

Sustainable Wood Supply

Timber-harvesting commitments have been made in the Little Smoky River area by the Alberta government. These commitments will be met by using a variety of innovative methods. The fundamental premise in the timber-harvest planning and approaches used will be to follow natural disturbance regimes as closely as possible. Wildfire has historically been the main influencing natural disturbance. Historically, wildfires vary in size, location, intensity, and reoccurrence cycles. Attempts will be made to understand these variations and to harvest the timber in a way that comes as close as possible to replicating fires across the landscape. To date, a study has been established that looks at how lichen regeneration was affected in a large harvested area and the effect on caribou–predator relationships on large cut over. A second study is under development that will look at selective harvesting and commercial thinning in lodgepole pine in an attempt to minimize visual disturbance and replicate a less intense wildfire.

Due to issues raised by local fishers and outside recreationists, efforts will be made to minimize the visual impact of timber harvesting. Such things as protection of roadside vegetation, small blocks, and non-clear-up systems will be considered in areas where there is potential for high recreation use.

Access

Development of a road network is critical to resource extraction, forest renewal, forest protection, and some recreation uses. The amount and type of roads developed, however, will have a marked effect on the inherent wilderness characteristics of the area. How roads are planned and constructed will be a very important aspect of the strategy. The first step proposed is identification of road corridors. The corridors will be identified on maps as locations for road construction when and if they are ever needed. Decision criteria for where roads should go will be based on all resource values. By discussing, preplanning, and integrating all concerns well in advance of actual road construction, the potential for conflict will be much reduced. The success of this approach will hinge on a commitment from all parties to develop the road plan together and then to follow it once it is complete.

Road-use controls will also be considered. Timing restrictions, for instance, to minimize sensory disturbance to wildlife during critical periods will be implemented where appropriate. Road closures will also occur both on a temporary, seasonal basis and once a specific road is no longer needed. Roads will be built to a minimum standard in terms of width, in the interest of minimizing disturbance while ensuring environmental integrity and user safety.

Fishery Protection

The Little Smoky River and its tributaries within the study area will be protected by a variety of means. In terms of harvesting activities, ANC Timber Ltd.'s operating ground rules define specific operating restrictions to ensure protection of watercourses.

In addition to ground rule controls, ANC is participating in research work to improve the level of understanding of the fisheries within the Little Smoky drainage. Studies are currently underway to determine the timing and extent of spawning migration of Arctic grayling. The results of this work will be used to direct activities within the Little Smoky area in a way that will not negatively affect grayling.

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Hydrogeology of brook trout (*Salvelinus fontinalis*) spawning and incubation habitats: implications for forestry and land use development

R. Allen Curry and Kevin J. Devito

Abstract: We demonstrated that nearshore spawning and incubation habitats of brook trout (*Salvelinus fontinalis*) are manifestations of lenses of coarse overburden materials underlying the nearshore zone. Lenses directed and accelerated groundwater flow into the habitats. They were <17 m wide, >1 m thick, and could be restricted to the nearshore zone or extend at least 20 m into the terrestrial catchment. Recharge areas necessary to sustain discharge in the habitats were estimated to encompass 3–10 ha, or 1–97%, of the associated terrestrial catchment. A 90-m buffer zone adjacent to the shoreline protected only 9–55% of the required recharge area. A hydrological approach to defining habitat protection measures is suggested.

Résumé : Nous avons démontré que les habitats de frai et d'incubation de l'omble de fontaine (*Salvelinus fontinalis*) sont des fonds où se retrouvent des accumulations lenticulaires de matériaux bruts, près des rives. Les lentilles de matériaux bruts dirigeaient et accéléraient le débit de l'eau dans les habitats. Elles avaient une largeur de <17 m, une épaisseur de >1 m et se retrouvaient soit près des rives, ou jusqu'à 20 m à l'intérieur de la zone terrestre du bassin hydrographique. Nous avons estimé les aires de recharge nécessaires au maintien de la décharge comme étant de 3 à 10 ha, soit 1 à 97% du bassin hydrographique terrestre associé. Une zone tampon de 90 m adjacente aux rives protégeait seulement 9 à 55% de l'aire de recharge requise. Nous suggérons une approche hydrologique pour définir les mesures de protection de l'habitat.

[Traduit par la Rédaction]

Introduction

The spawning and incubation of brook trout (*Salvelinus fontinalis*) in Canadian Shield water requires areas of distinct and constant groundwater discharge located in the nearshore zone (Curry and Noakes 1995; Curry et al. 1995). Known sites for reproduction within a water body are rare (<3), small in area (<10 m^2), and composed of unconsolidated, cobble–gravel–sand complexes (Snucins et al. 1992) created by glaciofluvial activities 10 000 – 13 000 years ago in this region (Price 1977).

The presence of shallow, unconsolidated, permeable materials overlying metamorphic bedrock suggests that groundwater is derived from local sources (Toth 1963). Local groundwater generated in the terrestrial catchment flows towards the lower potentials in the nearshore zone at a rate controlled by the hydraulic gradient and material permeability (Freeze and Cherry 1979). The greatest rate of discharge to surface water bodies occurs in the nearshore zone (Winter 1976). Given a uniform composition of overburden in the watershed, the rate of groundwater discharge would be similarly uniform across all nearshore areas. However, rates are extremely variable (e.g., Shaw and Prepas 1989) and they are significantly greater in brook trout reproductive habitats than in nearshore areas not selected by trout (Curry and Noakes 1995).

Despite the importance of groundwater discharge for successful brook trout reproduction, there is no information on the hydrological linkage between the required nearshore habitats and catchment where groundwater originates. The objective of this study was to increase our understanding of the interaction between catchment hydrogeology and brook trout spawning and incubation habitats. We hypothesized that lenses of coarse overburden materials accelerated the groundwater flow observed in the nearshore habitats. In addition, discharge rates were used to estimate the area of recharge necessary to sustain groundwater flow to the nearshore habitats and provide an evaluation of buffer zones as protection for brook trout spawning and incubation habitats.

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Methods

Study sites

Eight brook trout spawning and incubation sites in two lakes and one stream on the Precambrian Shield in central Ontario were examined from September to May, 1990–1992. The

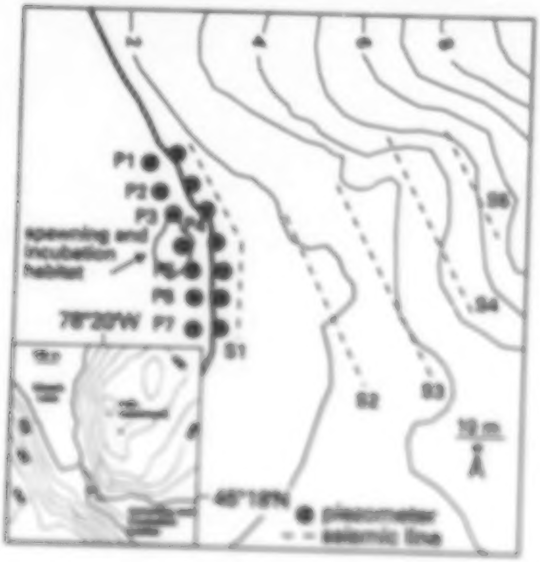


Fig. 1. Study area and terrestrial subcatchment associated with the brook trout spawning and incubation habitat at Meach Lake.

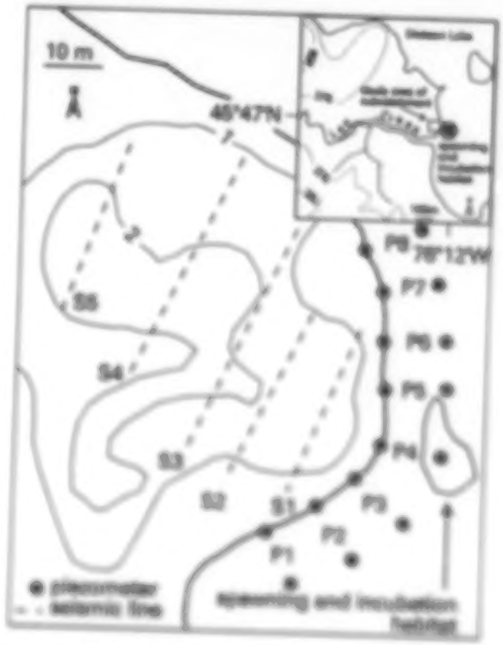
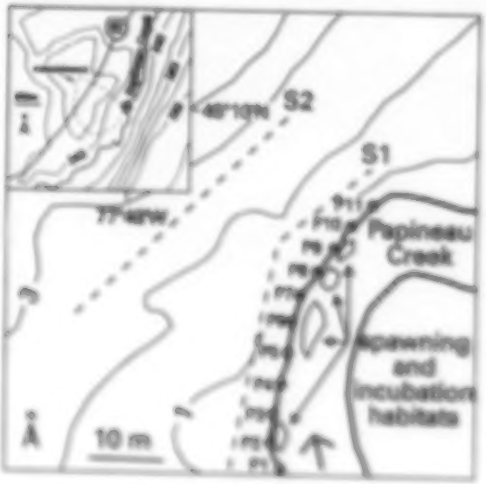


Fig. 2. Study area and terrestrial subcatchment associated with the brook trout spawning and incubation habitat at Dickson Lake.

Fig. 3. Study area and terrestrial subcatchment associated with the brook trout spawning and incubation habitat at Papineau Creek.



Sampling occurred on four or five occasions from August to November, February if surface ice did not engulf piezometers, and one or two occasions from April to June, 1990–1992. There was little variability in patterns among these samples (Curry 1993); therefore, the reported values are from samples taken in 1991 in early November in the lakes and late October in Papineau Creek when trout spawning was at its peak. All piezometers at one site were sampled on a single day. The hydraulic head differentials between the piezometer tip and the surface water (millimetres) were recorded. Vertical hydraulic conductivities were estimated following the methods of Hvorslev (1951). Onshore cross sections along each series of piezometer nests were generated by [illegible] (Golden Software, Inc., Golden, Colorado 1987) using the ordinary kriging technique [illegible] over the entire section.

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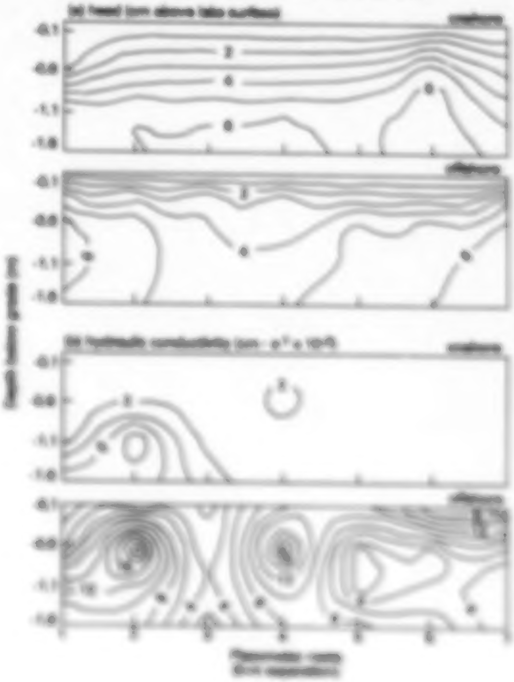
Fig. 4. Hydraulic head above the lake surface (*a*) and hydraulic conductivities (*b*) along the shoreline and through the reproductive habitat at Meach Lake.



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Results

Meach Lake

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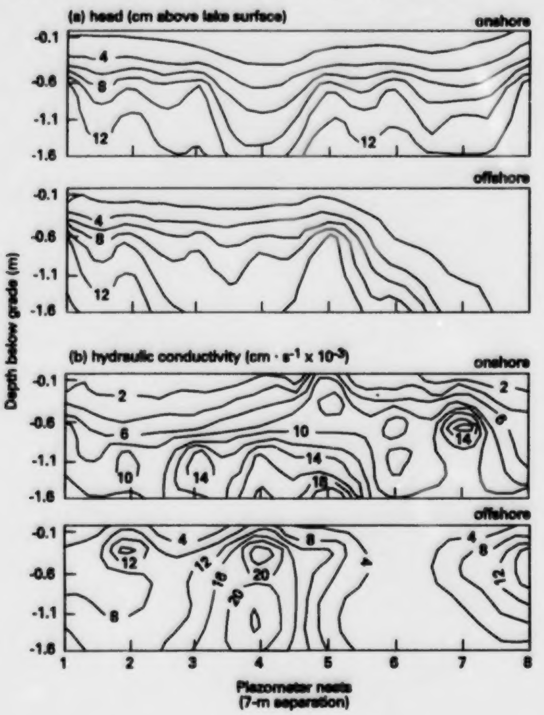

Fig. 5. Hydraulic head above the lake surface (*a*) and hydraulic conductivities (*b*) along the shoreline and through the reproductive habitat at Dickson Lake.

zone. Saturated sand 1.7–3.3 m thick dominated (1136 $m{\cdot}s^{-1}$) this area with possible gravel or cobbles (1395 $m{\cdot}s^{-1}$) in the vicinity of P2. Distinct pathways of flow from the terrestrial to the aquatic portion of the catchment were not apparent.

The mean discharge of groundwater in the reproductive habitats was 1.6×10^{-3} $cm{\cdot}s^{-1}$, or 1.2×10^{7} L of groundwater discharging annually.

Dickson Lake

The hydraulic heads in the nearshore zone at Dickson Lake indicated groundwater was discharging to the surface waters offshore (Fig. 5*a*). Flow converged from the north and south towards P4 and P7 along the shoreline. Offshore, there was a convergence at P3 and P4 with no flow apparent at P8. The hydraulic conductivities along the shoreline were greatest below 60 cm from P3–P5 and P7 (Fig. 5*b*). Conductivities increased offshore particularly at the substrate surface between P2 and P5. Conductivities also increased in the shallow substrate of P7 and P8. A pathway of flow approximately 14 m wide was suggested at P3–P5.

The thickness of the overburden material was estimated to be >7 m (no bedrock arrivals from the seismic survey were recorded). The unsaturated zone 2.3–3.2 m thick and composed of heterogeneously distributed unconsolidated sand, gravel, cobbles, and boulders (270–335 $m{\cdot}s^{-1}$). The saturated zone was composed of sand, gravel, and cobbles

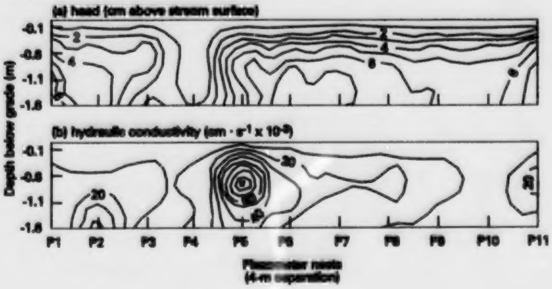

Fig. 6. Hydraulic head above the stream surface (*a*) and hydraulic conductivities (*b*) along the shoreline at Papineau Creek.

(1530–2696 $m{\cdot}s^{-1}$). The coarsest materials were located in the 30 m adjacent to the shoreline in the southwest (S1–S3).

The mean discharge of groundwater in the reproductive habitats was 1.7×10^{-3} $cm{\cdot}s^{-1}$, or 4.1×10^{7} L of groundwater discharging annually.

Papineau Creek

Hydraulic heads suggested an upwelling of groundwater towards the surface water dispersing from areas at P2 and P6–P10 (Fig. 6). Hydraulic conductivities suggested a primary lens of coarser materials 16 m wide and between 25 and 175 cm deep extending from P5 to P9 (Fig. 6). A second lens of coarser materials was apparent >100 cm deep at P2.

The thickness of the overburden material was estimated to be >10 m (no bedrock arrivals from the seismic survey were recorded). The unsaturated zone in the terrestrial catchment was 1.6–2.4 m thick and consisted primarily of sand (327–581 $m{\cdot}s^{-1}$). The saturated zone was composed of primarily sand (1532–1673 $m{\cdot}s^{-1}$). Coarser material (1737–1829 $m{\cdot}s^{-1}$) was located in the 20 m adjacent to the stream in the south (S1).

The mean discharge of groundwater in the reproductive habitats was 1.9×10^{-3} $cm{\cdot}s^{-1}$, or 2.6×10^{7} L of groundwater discharging annually.

Discussion

Successful brook trout reproduction in Canadian Shield waters requires coarse substrate materials <1 m thick with distinct and constant upwelling groundwater (Curry and Noakes 1995; Curry et al. 1995). Our observations indicate these habitats are manifestations of lenses of coarse materials that underlie the nearshore zone. The lenses can be restricted to the nearshore zone (Meach Lake), or extend at least 20 m into the terrestrial catchment (Dickson Lake and Papineau Creek). They displayed varied dimensions of >1 m thick and 10–16 m in width along the shoreline, sometimes with smaller, adjacent projections <5 m wide. Each lens directed groundwater flow into the spawning and incubation habitats at flow rates greater than adjacent nearshore areas (Curry and Noakes 1995).

Using the observed groundwater discharge and annual precipitation inputs, it is possible to estimate an area of recharge necessary for sustaining groundwater travelling through lenses and into the spawning and incubation habitats. Studies of hill-slope hydrology in similar catchments in humid climates indicate only a portion of precipitation recharges shallow, local groundwater with most lost to evapotranspiration and through quick-flow pathways (Devito et al. 1996; McDonnell and Taylor 1987). Using a conservative annual estimate of 100 cm precipitation and 40% recharge, the estimated area of recharge necessary to sustain the observed groundwater discharge is 3.0, 6.4, and 10.3 ha of the terrestrial subcatchment areas at Meach Lake, Papineau Creek, and Dickson Lake, respectively. Consequently, up to 97% (Meach Lake) or 10 ha (Dickson Lake) of the terrestrial catchment appears directly linked to brook trout spawning and incubation habitats.

Lenses are most likely buried glaciofluvial deposits (White 1974) such as eskers (possibly Dickson Lake), or remnants of glacial stream channels (Papineau Creek and Meach Lake). Reproductive habitats would be created when postglacial changes in surface water levels or erosion (glaciofluvial or present day channel cutting, e.g., Papineau Creek) exposed the lenses in the nearshore zone. This dynamic nature of groundwater development and pathways within glacial deposits indicates that the time when a lens could become available to brook trout and the duration of the lens's ability to support reproduction are variable and unknown. It is common in Canadian Shield waters for trout to maintain secondary spawning sites where there is significantly reduced activity or sporadic usage (R.A. Curry, unpublished data). Secondary sites may be new, developing, or older, deteriorating areas of discharging groundwater that represent the trout's evolutionary adaptation to the dynamic groundwater systems on which they depend.

A link between groundwater development and pathways in terrestrial catchments and brook trout reproductive habitats has critical implications for integrating sustainable forestry and fishery management. Increasing groundwater levels following removal of the forest cover (Peck and Williamson 1987) may enhance groundwater delivery to the nearshore zone and therefore the quality of spawning and incubation habitats. Alternatively, a shallower water table could alter the temperature of groundwater delivered to the nearshore habitats and affect incubation success (Hokanson et al. 1973). Successful incubation also requires a stable environment (Marten 1992). Groundwater temperature, chemistry, and flow are stable in spawning and incubation habitats (Curry et al. 1995). Stability may be jeopardized by fluctuating rates of groundwater discharge that occur after timber harvesting (Wright et al. 1990).

Preservation of brook trout spawning and incubation habitats during timber harvesting typically focuses on the direct protection of the nearshore zone. For example, a buffer zone or reserve adjacent to the shoreline can be created, e.g., ≤90 m (Ontario Ministry of Natural Resources 1988). A 90-m buffer zone would protect <9, <23, and <55% of the recharge areas required to sustain the reproductive habitats at Dickson Lake, Meach Lake, and Papineau Creek, respectively, assuming the critical areas for recharge are located adjacent to the nearshore zone. Such buffer zones may be inadequate if they do not encompass and protect the entire recharge zone associated with the discharge zone being used by trout. It would be more realistic to view the spawning and incubation habitats in terms of necessary hydrological requirements and thus develop a definition of a buffer zone in terms of hydrological units.

Other development activities within a catchment such as creation of reservoirs, removal of aggregate materials, and the construction of roads and human residences may alter groundwater recharge and impact spawning and incubation habitats. Development can also affect groundwater quality. Curry et al. (1993) observed road salt contamination of the groundwater in brook trout spawning and incubation habitats. Nutrients can also be introduced to groundwater discharging in the nearshore zone from human septic systems (Lee 1972) and agriculture (Peterjohn and Correll 1984). Protection from these activities will also require the recognition of the hydrological characteristics of spawning and incubation habitats.

Our evidence indicates a linkage exists between brook trout reproduction and catchment hydrogeology. There are clearly questions that remain unanswered regarding the specific hydrological relationships, as well as the impacts of land use on groundwater and brook trout reproduction. Nonetheless, present-day protection of habitats does not incorporate the hydrogeological characteristics of the spawning and incubation habitats. Protection guidelines must be reviewed to achieve the successful integration of terrestrial and aquatic resource management in watersheds inhabited by brook trout.

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Estimating the Cumulative Long-term Effects of Forest Harvests on Annual Water Yield in Alberta

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Abstract

Our purpose in this paper is to demonstrate the capability of a data-based version of the United States Environmental Protection Agency's Water Resources Evaluation of Non-point Silvicultural Sources procedure to estimate the hydrologic effects of various forest harvesting practices in Alberta. We use our WRNSSDR program, coupled with the results of experiments and studies done on Forest Management Agreement areas in Alberta over the past 25 years on the rate of annual water yield change in the years following harvest, to estimate the cumulative effect of several harvesting sequences on the magnitude and duration of increased annual yield. The Alberta Forest Service's Phase III inventory data are used to derive functions for simulating regrowth on clearcuts. In coniferous forests the simulations indicate that annual water yield will be increased substantially for about 50 years. In deciduous forests, yield increases, which may be initially quite large, are estimated to vanish in fewer than 30 years. Simulations of intensive clearcutting on small boreal watersheds indicate the potential of utilizing forest harvest to emulate fire or catastrophic insect attacks to mimic their effects on annual streamflow.

Swanson, R.H., Wynes, R.D., and Rothwell, R.L. 1998. Estimating the cumulative long-term effects of forest harvests on annual water yield in Alberta. Pages 83–93 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Forest Harvesting Interactions with Fisheries

Forest harvesting modifies the flow regime of a stream (Anderson et al. 1976). But it is difficult to judge the worth of an alteration in flow regime because one cannot state *a priori* that an increase in water yield is good, bad, or inconsequential. Each stream is an ecosystem and an effect on its flow regime may be insignificant, beneficial, or detrimental, depending upon which organisms are affected and how, when, and for how long changes in streamflow occur.

Because forest harvesting affects streamflow, it should be desirable to set water-oriented goals for forest management activities. The establishment of such goals requires the input of those concerned with water supply, both as it relates to downstream users and to stream inhabitants. For instance, on flood-prone streams in Alberta, forest harvest on their watersheds should not increase annual water yield by more than of 15% of average annual water yield (John Taggart, Alberta Environmental Protection, Edmonton, Alberta, personal communication). A 15% increase in annual water yield was considered to have an insignificant affect on streamflow peaks. Similar goals or objectives for the management of forest harvesting activities to enhance or maintain aquatic habitat for fish are non-existent.

Forest harvest is often compared to the natural removal of forest cover by fire, insects, or diseases. Both fires and catastrophic insect or disease attacks have the potential to kill trees over vast areas. Dead trees do not use water but do provide some shade and wind protection. To some extent forest harvest emulates these natural events. Harvesting eliminates water use by the trees removed, but it also eliminates much of the shade and wind protection until regrowth occurs and a new stand is established. The recurrence interval of natural events is unplanned, but may be similar in duration to that of planned sequences of harvest (rotation). The areal extent and intensity of natural events are largely uncontrolled, although it is axiomatic among foresters to reduce the probability of catastrophic events by manipulating the age and structure of a stand. Under forest management, scheduling time and intensity of harvests presents an opportunity to purposefully modify water resources, either to emulate nature, or to achieve water-oriented goals.

Our purpose in this paper is to demonstrate the capability of an existing procedure to estimate the hydrologic effects of various forest harvesting practices in Alberta on the cumulative magnitude and duration of annual water yield changes. This procedure can also be used to simulate natural events so that they can be compared with forest harvest. We use our WRNSSDR program, (the hydrology portion of the United States Environmental Protection Agency's WRENSS procedure for snow-dominated regions, United States Environmental Protection Agency 1980), coupled with the results of experiments and studies done on Forest Management Agreement (FMA) areas in Alberta over the past 25 years on the rate of annual water yield change in the years following harvest, to estimate the cumulative effect of several harvesting sequences on the magnitude and duration of increased annual yield.

The WRENSS Hydrologic Procedure

The United States Environmental Protection Agency handbook *Water Resources Evaluation of Non-Point Silvicultural Sources: A Procedural Handbook* (WRENSS) contains several procedures for estimating changes in water resources that should be expected following forest harvest (United States Environmental Protection Agency 1980). The hydrology portion of WRENSS is used to calculate an estimate of local evapotranspiration as a function of seasonal precipitation, vegetative cover density, and climatic region. Annual water yield is determined by subtracting the derived evapotranspiration from seasonal precipitation and summing the seasonal water yields. WRENSS calculates annual generated runoff[1] only and does not carry annual surpluses or deficits to subsequent years.

Chapter III, the hydrology portion of the WRENSS handbook, contains a set of graphs that one can use to estimate the annual water yield changes associated with forest harvest within a climatic region. One set is intended for use in rain-dominated areas where the pattern of runoff (hydrograph) responds quickly to precipitation events throughout the year (Fig. 1), and a second set for snow-dominated areas where the hydrograph displays little or no response to precipitation during a winter season followed by a primary or dominant flow event during a snowmelt season (often accompanied by significant rain) with secondary responses

[1] Generated runoff is defined as precipitation minus all of the losses that will occur. It is water that will eventually become streamflow, but may be retained in storage within the watershed or stream channel for some indefinite period before it flows past the watershed boundary (United States Army 1956).

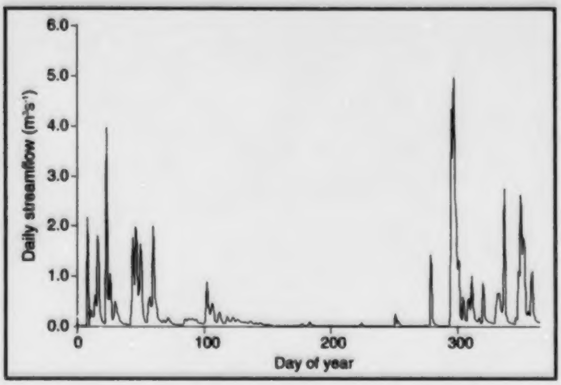

Figure 1. Hydrograph from a rain-dominated watershed.

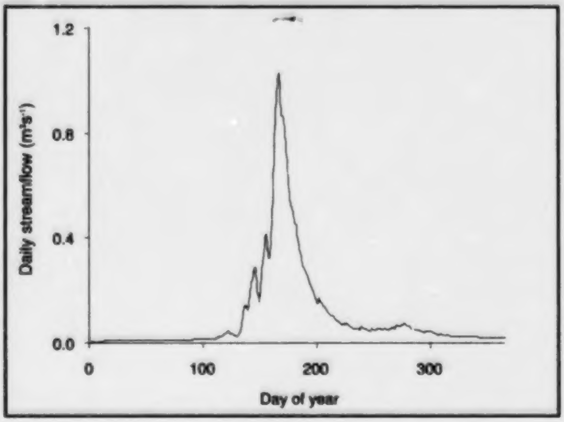

Figure 2. Hydrograph from a snow-dominated watershed.

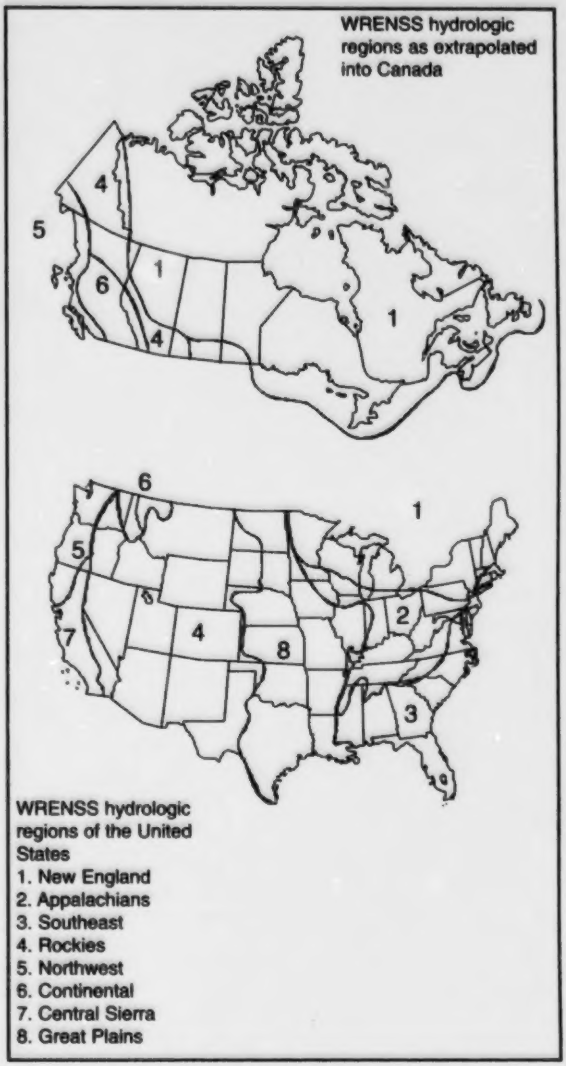

Figure 3. The WRENSS hydrologic regions of the United States and Canada.

to summer precipitation events (Figs. 2 and 5). The United States was divided into eight climatic regions (Fig. 3). These climatic regions have been extrapolated into Canada based on the forest associations dominant in each region.

The WRENSS graphical procedure was time-consuming to use and has apparently been little used in either the United States or Canada. The Northern Forestry Centre, Canadian Forest Service, Edmonton, Alberta, produced a computer version for MS-DOS (WRNSHYD version 1.0) of the hydrological procedures, which was distributed in 1989. Development and refinements to WRNSHYD ceased with version 1.1 in 1991, and it is no longer supported or supplied by the Canadian Forest Service.[2] The WRNSHYD program was written in Borland Turbo Pascal™ with a binary data format, which is difficult to access except from within the WRNSHYD program. This non-standard data format limited the use of WRNSHYD to manual entry of a few data sets. WRNSHYD has no provision for regrowth to occur as regeneration of a clearcut takes place, although one can enter basal areas and tree heights manually.

[2] The MS-DOS WRNSHYD version 1.1 is still available free of charge, except for $5.00 media and mailing costs, from RH Swanson & Associates, #28, 216 Three Sisters Drive, Canmore, Alberta, T1W 2M2 (403) 678-6096.

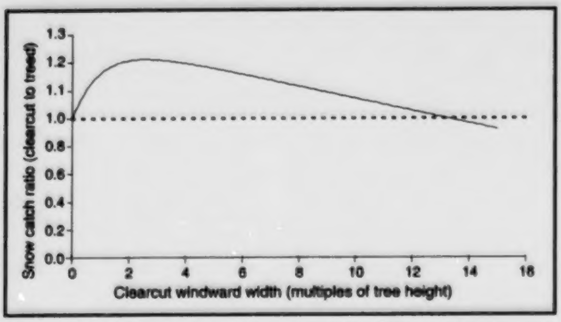

Figure 4. Snow retention as a function of clearcut windward dimension ≤13 tree heights across. Data for clearcuts ≤6 tree heights in width from Golding and Swanson (1978); for clearcuts >6 tree heights in width from United States Environmental Protection Agency (1980).

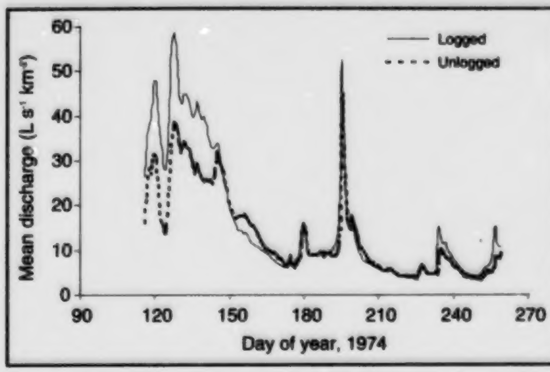

Figure 5. Hydrographs from two composite watersheds, one consisting of nine partially harvested and the other of nine unlogged catchments near Hinton, Alberta (Swanson and Hillman 1977).

Our current development of computer versions of the WRENSS hydrologic procedures is toward interfacing its hydrologic routines with a database. This will allow one to utilize input data from a variety of sources, such as a Geographical Information System (GIS), or to retain and easily edit input data from previous simulation runs. It will allow results to be retained in a format suitable for further analysis or incorporation into descriptive material accompanying an geographical entity within a GIS. The most recent version is WRNSSDR,[3] a Microsoft ACCESS™ database implementation of the WRENSS procedure for snow-dominated regions only. The WRNSSDR program allows the user to provide for tree height and basal area growth on clearcuts as functions of time since harvest, thus enabling the user to simulate the effect of a single harvest or the cumulative effects of a sequence of harvests through one or more forest rotations. The WRNSSDR program was used to produce the simulations for this paper. Except for the database orientation and the incorporation of regrowth, WRNSSDR is identical in structure to the snow-dominated routines in WRNSHYD. The WRNSSDR program is entirely metric and does not perform conversion from Imperial to metric units, or vice versa as does WRNSHYD.

Two features of the WRENSS procedure set it apart from most hydrologic models: 1) it requires very little data to operate, and 2) it incorporates the spatial distribution and redistribution of snow.

The only real data required to operate WRENSS is monthly or seasonal precipitation for the area in question. Harvest and stand data for the specific site being evaluated are desirable, but one can make reasonable estimates of these for a preliminary simulation of estimated harvesting effects on annual water yield.

The second feature is the areal distribution of snow as affected by spatially and temporally discontinuous forest harvesting, i.e., clearcut blocks created at various times and areally distributed over the landscape. Most of us have observed that snow accumulation in clearings is different from that under the uncut forest. Four physical processes account for the differential amounts of snow found in clearings versus the uncut forest: 1) lack of interception, 2) wind turbulence at the edges of clearings and treed areas, 3) redistribution of snow initially caught in the canopy of trees adjacent to clearings, and 4) erosion, transport, redeposition, and sublimation of snow from the snowpack within a clearing (Anderson et al. 1976; United States Environmental Protection Agency 1980; Troendle et al. 1988). The first three processes act to increase the amount of snow in a clearing. The removal of the canopy eliminates interception, allowing all of the falling snow to reach the surface of the clearing. Wind speed in a clearing is lower than above the canopy. This slower speed reduces the capacity of the air to carry snow and enhances its deposition in a clearing. Wind

[3] The WRNSSDR program is commercial software developed by RH Swanson & Associates and is not available for free distribution. It is currently (September 1996) in the beta testing phase.

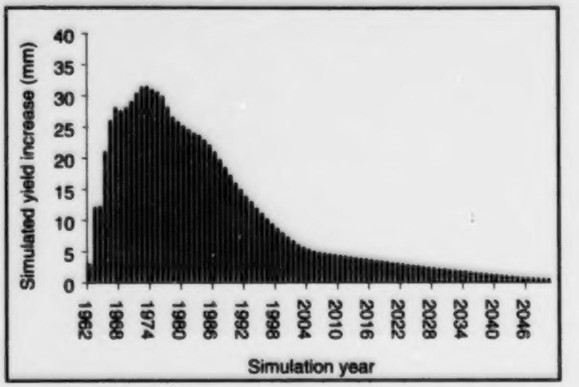

Figure 6. The WRNSSDR simulation harvest effects in lodgepole pine and spruce–fir forests on water yield in the area around Hinton, Alberta, fair site quality (height 8–10 m at 50 years breast-height age).

movement through the canopy of the trees surrounding a clearing is thought to transport some of the intercepted snow to the clearing (United States Environmental Protection Agency 1980).

The last process acts on the snowpack to reduce the amount of snow in clearings. Wind can detach particles of snow and physically transport it to another location. Some evaporation (sublimation) can occur during such transport (Tabler 1975). Sublimation from the snowpack surface will also reduce the amount available to melt in the spring. In WRNSSDR, erosion and *in situ* sublimation are both wind-driven functions of wind speed at the snow surface, which is dependent upon the height of regrowth and clearcut dimensions.

Incorporating Alberta Research Findings

In all versions of WRENSS, snow is distributed between treed and cleared areas as a function of clearing size. In WRNSHYD and in WRNSSDR, in clearings smaller than 13 tree heights in the windward direction, an empirical relationship (Fig. 4), derived from Alberta data (Golding and Swanson 1978), is used to apportion the snowpack between cleared and treed areas. In clearings where the windward dimension is greater than 13 tree heights, some rather complex equations are used to apportion the snowpack and to account for sublimation of windblown snow during transport within the clearing and the deposition of surviving snow particles into downwind treed areas (pages III.148 – III.152 *in* U.S. Environmental Protection Agency 1980; Tabler 1975).

Tabler (1975) provided the option of using a constant value for over-winter *in situ* sublimation from the snowpack or of simply ignoring it in his equations. Snow loss via *in situ* sublimation is principally driven by vapor pressure deficit and wind speed (Satterlund 1972) and can be an important hydrologic component of water yield from clearings in forested watersheds in Alberta. *In situ* sublimation is computed as an optional routine in WRNSSDR, applicable to the snow in all clearings, not just those greater than 13 tree heights across in the windward direction. The amount of sublimation that occurs during the winter is a function of the wind speed in an opening and the number of days that it does not snow during the winter. Sublimation is assumed to be zero on days that it does snow because the vapor pressure difference between the snow surface and the atmosphere would be very small or zero.

Regrowth Effects on Evapotranspiration

Regrowth of vegetation affects evapotranspiration in WRENSS in two ways: 1) through basal area increases, which affect the cover density on a site, and 2) through height growth, which affects snow redistribution and loss in transport. In WRNSSDR, regrowth also affects the wind at the surface of a clearing and the *in situ* sublimation from the snowpack.

Immediately after clearcutting, ET in WRENSS is reduced to 30–70% of that prior to harvest, i.e., at cover density = 0 (United States Environmental Protection Agency 1980). As regrowth occurs, evapotranspiration returns to pre-harvest levels when the cover density is at approximately 50% of that prior to harvest (United States Environmental Protection Agency 1980). It therefore follows that the rate of regrowth, particularly during the first few years after clearcutting, is critical to the recovery of evapotranspiration to pre-harvest levels.

The cover density of the vegetation in clearcuts and treed areas, in the snow-dominated routines of WRENSS, is a computer-derived term indexed to basal area (United States Environmental Protection Agency 1980). We have incorporated provision for the entry of user-defined functions that increase the basal area in a clearcut, with time since harvest, into WRNSSDR. There are two features that these functions must possess to be realistic: a shape that accurately depicts tree growth with time, and a realistic maturity age. Leaf and Brink (1975) suggested the following maturity ages for Rocky Mountain forest species: lodgepole pine–basal area 80, tree height 120 years; spruce–fir–basal area 100, tree height

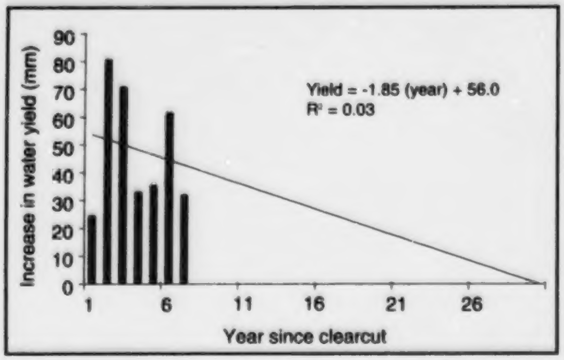

Figure 7. Rate of change of water yield increase following aspen harvest on the Streeter Basin experimental watershed, Alberta (Swanson et al. 1986). The bars represent actual water yield increases. The line is the linear regression of the actual increases extrapolated to the year (30) when the water yield increase are zero.

160 years; and aspen–basal area 60, tree height 80 years. Unless otherwise specified, we have derived the shape of basal area and tree height functions from Alberta Forest Service Phase III inventory tables and use the ages cited above as boundary conditions for basal area and tree height for all simulations reported in this paper.

Basal area was used in WRENSS because it is generally available for all managed forest stands, it is relatively easy to measure, and is a good index of cover density in mature stands. We feel that it is less so in regenerating stands because basal areas are generally taken for commercially important species only, and at an age when it can be measured at breast height (1.15 m). We also feel that basal area is not as good an index of cover density in second growth stands because it can return to pre-harvest levels before full re-occupancy of the soil by functioning roots.[4]

From a hydrologic standpoint, all vegetation has some importance in determining evapotranspiration from a site. Better indices of cover density, such as a combination of rooting depth and leaf area index or biomass of all species on the site, could be incorporated into WRNSSDR, but such information is not available from existing forest inventory data.

Regrowth Effects on Snow Distribution

Height growth affects snow disposition and distribution in clearings with windward widths greater than 13 tree heights (United States Environmental Protection Agency 1980). The effects of regrowth on snow distribution are evident until the height of the trees in a clearing are equal to those of the surrounding stand. The effect of differential height on snow distribution is not expected to vanish for about 60 years in aspen stands, 120 years in lodgepole pine stands, and 160 years in spruce–fir stands (Leaf 1975; Leaf and Brink 1975).

Slash and regeneration height is used in the snow loss equations (Tabler 1975) to limit wind erosion from the surface of the snowpack. In WRNSSDR, we have assumed that when the height of the regenerating trees is 1 1/2 times the depth of the snowpack, it is protected from the wind and no further erosion or transport of snow occurs. Snow in transport is subject to sublimation loss. If an average-sized particle of snow travels approximately 1 km, it will completely sublimate (Tabler and Schmidt 1972). Transported snow that does not travel far enough to sublimate is deposited in the downwind uncut portion of the watershed (Tabler 1975). This erosion and redeposition process reduces the snowpack in large clearings (those with windward widths greater than 13 tree heights) and increases it to a lesser degree in the uncut portion of the watershed.

Verification of WRENSS Results

The simulated and actual water yield values for three Alberta watershed studies were used to test the accuracy of the WRNSSDR estimates (Table 1). In general, all procedures based on WRENSS can be expected to underestimate the effects of clearcutting on water yield because it was designed specifically to do so. (All graphs in the WRENSS handbook represent the lower 90% confidence interval of estimates of water yield change. C.F. Leaf, Merino, Colorado, personal communication.) This was done so that estimates of water yield increase would be conservative because the goal of most watershed management projects in the United States, at the time WRENSS was published, was to increase water supply. This bias towards conservative estimates of

[4] The rate of regrowth of aspen and other deciduous species in the Boreal forest, and its effect on evapotranspiration and subsequent water yield, is the subject of investigations started in 1992 by Forestry Canada, the University of Alberta, Daishowa-Marubeni International, Ltd., and Alberta Environmental Protection. Preliminary indications are that the combined basal area of all regrowth increases rapidly and approaches pre-harvest conditions within about 5 to 10 years. However, soil moisture measurements at 15 and 40 cm depths from the surface indicate that the roots of the regeneration vegetation do not fully occupy the soil at the 40 cm depth for 10 or more years.

Table 1. Comparison of WRNSSDR simulations of annual water yield increase and experimental results from watershed studies in Alberta

Watershed	Actual increase (mm)	Simulated increase (mm)
Marmot, Cabin Creek (Swanson et al. 1986)	17	11
Streeter Basin (Swanson et al. 1986)	81	82
18 catchment composite, Hinton (Swanson and Hillman 1977)	41	26 to 37[a]

[a] The value simulated depends upon the quality of the site that is used in WRNSSDR to estimate regeneration basal area and height regrowth with time since clearcut. On a good site (height 18 m at 50 years breast-height age) the simulated annual yield increase in 1974 is 26 mm; on a fair site (height 8–10 m at 50 years breast-height age), the simulated annual yield increase in 1974 is 37 mm.

water yield increases, which is built into WRENSS, probably has very little practical effect when one considers the normal variation in water yield increases that occur from year to year (for example, note the variation in the increases that occurred over the first 7 years at Streeter Basin, Alberta, Figure 7).

Simulation of the Cumulative Effects of Harvest Through Time

In this section we present some examples of how we have used the WRNSSDR program to examine the hydrologic effects of forest harvest on annual water yields. An actual study was chosen for simulation of the effect of coniferous harvest because we had data to compare with the simulations. The simulations of deciduous harvest effects are all hypothetical because of the uncertainty that currently exists with respect to the amount of regrowth required to achieve hydrologic recovery.[5] These are intended only to illustrate the type of information that can be derived with the WRNSSDR program and should not be considered as recommendations for forest management to achieve water-oriented goals.

Coniferous Forests

An evaluation of water yield from lodgepole pine and spruce–fir watersheds was conducted at Hinton, Alberta, in 1973 and 1974. In this study 18 separate watersheds, nine logged and nine unlogged, were gauged during part of 1973 and all of 1974. These watersheds were harvested at various times since the early 1960s, and by 1974, 53% of the area of the logged watersheds had been clearcut. The flows from the logged and unlogged watersheds were averaged and the mean unit-area annual yields compared (Fig. 5) (Swanson and Hillman 1977). The average increase in yield measured, 41 mm, may have been greater or smaller in prior years; this 1 year of measurement indicates nothing about past history or the future.

We used the WRNSSDR procedure and annual cutting data (Swanson and Hillman 1977, Table 2, page 21), to simulate the Hinton composite watersheds results for 80 years, starting with the year of the first clearcut (Fig. 6). Both height and basal area regrowth were estimated as functions of the number of years since clearcutting, with equations derived from Alberta Forest Service Phase III inventory tables. The average age (6–11 years) of the clearcut blocks on each watershed was used to calculate regrowth on each of the nine watersheds.

The simulated increase in yield with time (Fig. 6) is referenced to the water yield in the uncut state just prior to the first clearcut. The simulated increase in water yield from this sequence of harvest peaked in 1974. The decline in increase between 1967 and 1969 represents a reduction in the rate of harvest during those years (Fig. 6). Reentry to harvest the second growth was scheduled for 2041, 80 years after the initial harvest. The simulated yield increase in 2041

[5] Unpublished analysis of data from the Streeter Experimental Basin in southern Alberta. Here, aspen forest was harvested in 1976, and water yield data collected through 1982 (Swanson et al. 1986). The projected decline in water yield during the 7 years of post harvest record indicates that all effects of the harvest will vanish at year 30 ± approximately 15 years.

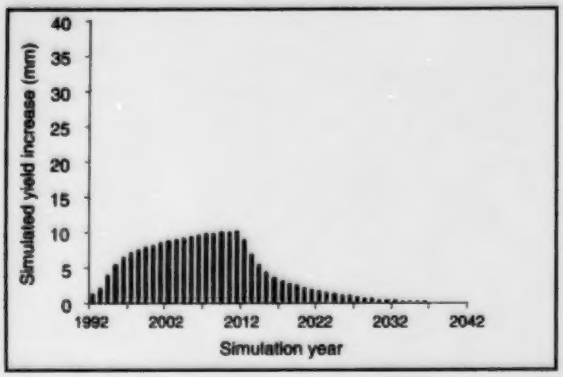

Figure 8. The WRNSSDR simulation indicates that rapid and high volume harvest results in minimal impact on water yield from a 355 km² deciduous watershed in northern Alberta.

indicates that a residual increase in water yield of approximately 2 to 3 mm will remain from the first harvesting sequence.

We do not know what effect this sustained water yield increase would have on aquatic habitat. Most of the increase measured in 1974 occurred during the spring freshet, although there are also lesser increases in the volume of flow from summer rainstorm events (Fig. 5). In order to use this simulated flow information, one would have to examine the historical hydrographs of streams in this area to see what flow volumes, if any, were correlated with fish habitat disturbance. Then one would need to superimpose simulated increases on the historical hydrographs to see if the frequency of flow events large enough to cause habitat disturbance would increase. The important point to remember is that under the current type of management used in coniferous-forested watersheds in this part of Alberta, for approximately 50 years after a first clearcut is created, or slightly more one-half the planned rotation period, streamflow during the spring freshet and the magnitude of peak flows from rain events, will likely be greater than if no harvest had occurred.

Deciduous Forests

We do not have the same historical base to use as a reference in evaluating the effects of sequential harvests in deciduous forests on water yield increases as we do for those of the foothills coniferous forests. Our only water yield data on the time required for hydrologic recovery of streamflow from deciduous forest harvest is that obtained from the Streeter Basin experiment in southern Alberta (Fig. 7), which indicates the hydrologic effect will vanish in approximately 30 years. Other less-direct data from our study of regeneration on aspen clearcuts in the Boreal forest region indicate that hydrologic recovery may occur faster than the Streeter Basin data indicate (see footnote 4).

Given the limitations on our understanding of the true rate of increase in cover density of deciduous stands following clearcutting, we have used the best estimates of basal area increase with time that are currently available to simulate hydrologic recovery, i.e., the Alberta Forest Service's Phase III inventory data. These data, from fire-origin stands, indicate that the current basal area of mature deciduous stands took 60 to 180 years to develop, depending on site quality. In the simulations of deciduous regrowth with WRNSSDR reported below, we used the height and basal area regrowth data from stands on good quality sites to derive basal area and tree height regrowth functions, i.e., site index 20 m at 50 years breast-height age (approximately 60 years stump age). We used the data from the Streeter Basin study to set full hydrologic recovery at 30 years (Figure 5), a value that appears to be conservative in that the recovery time is probably even shorter.

For illustrative purposes in this paper, we simulated the sequential harvest and regrowth of aspen stands on the 667 km² Keg River watershed in northwestern Alberta. We assumed that these stands will continue to be managed for aspen. The Keg River has been identified as flood-prone, and Alberta Environmental Protection has indicated that increases in average annual yield of 15% or less should not adversely impact downstream users (in this case, farmlands on the floodplain of the Keg River). For the initial harvesting sequence, we were constrained by both the 15% criterion, and by a time constraint because the timber is mature, and the quality of the aspen stand will likely start to deteriorate within the next 30 years. Subsequent harvesting sequences in the second rotation will not be subject to the same 30-year time constraint. Although some harvest had already occurred since 1991, the majority of the timber within the watershed remains to be harvested over the next 25 years. Therefore, areas harvested in 1991 through 1993 are actual; all those in subsequent years are hypothetical, as both the total area harvested each year and the location of that harvest within the watershed are subject to change as operating conditions change.

The area of the Keg River watershed area above the first affected downstream user is 355 km². About

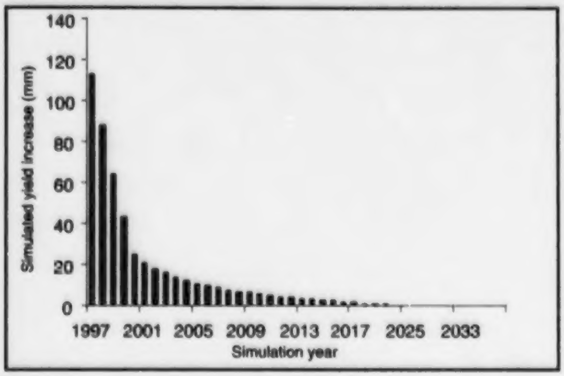

Figure 9. The WRNSSDR simulation of the increase in water yield caused by harvestinq 80% of a 5 km^2, aspen-forested watershed, in 1 year.

300 ha were harvested from this watershed in each of 1991/92 and 1992/93, and 250 ha in 1993/94. Various combinations of annual harvests from 1994/95 through 2020/21 were suggested as possibilities for simulation. The most drastic, harvesting the maximum volume possible within the current operating limitations of the pulp mill of 620 ha per year, resulted in a simulated increase of 10.2 mm or 9.2% at the first downstream user (Fig. 8).

Scale of Operations

When considering the effects of forest harvest on water users, we must consider the areal scale of the forestry operations. On a relatively large watershed, such as the Keg River, the simulated impact of normal harvest operations on downstream users is relatively small. However, on smaller watersheds that are similar in size to the area that would be harvested in a given year, the magnitude of the possible water yield change and the potential for managing or mismanaging impacts are much greater. For example, in the Boreal forest, the watershed area of identifiable first- or second-order streams is about 400 to 1000 ha. Assume that a pulp mill requires the volume of wood from 400 to 600 ha for 1 year's operation. On these small first- or second-order watersheds, it would be operationally possible to remove all of the merchantable area in 1 year. These watersheds provide additional possibilities for "managing" the rate of forest harvest to achieve specific water yield objectives.

For instance, consider a 500 ha watershed with an average annual water yield of 110 mm, and 400 ha of operable and merchantable deciduous timber. The simulated increase in water yield from harvesting 400 ha in 1 year is 112 mm (Fig. 9), more than double the average annual water yield. It may be unlikely that harvesting of this intensity would be applied to a watershed of this size in the Boreal region. However, if the goal of the harvest was to produce the maximum possible effect on water yield, then our simulations indicate that intensive harvesting in these small watersheds would achieve it. Another simulation (not shown) also indicates that a similar increase in yield could occur if a catastrophic fire or tree-killing insect attack killed the trees on 80% of the area of any size of watershed.

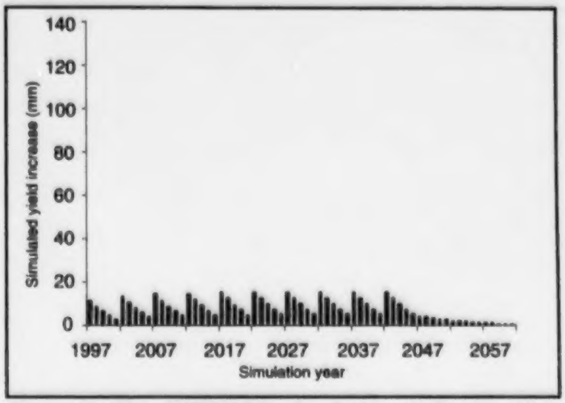

Figure 10. The WRNSSDR simulation indicates that harvesting of 40 ha every 5 years on a 5 km^2 aspen-forested watershed regulates water yield, and has the potential to both increase and to maintain water yield at a higher level than prior to harvest.

The harvest on this 500-ha watershed example could be designed to produce a more stable effect on water yield. Consider a 50-year rotation, and an annual harvest of 40 ha, spaced at 5-year intervals, until all 400 ha of merchantable timber was removed. Our simulations indicate that this harvesting regime would increase annual water yield until it stabilized at 15.5 mm or 14% of average annual yield (Fig. 10), and since the last harvest occurs 45 years after the first, the block harvested in 1997 would be ready to harvest again in 2047. If this second and subsequent harvests occur, the average annual water yield from this watershed would be increased from 110 mm to a new level of about 120 mm in perpetuity.

The examples above represent only two harvesting scenarios out of many possibilities that could be designed and simulated to meet water yield objectives. The WRNSSDR program has the capability of

handling any number of harvesting and/or catastrophic scenarios. The area clearcut each year and the time between entries into a watershed can be modified in any combination consistent with harvesting constraints to simulate effects on water yield.

Discussion and Conclusions

Annual Variability

As with any modeling technique, the simulations produced by WRNSSDR should be viewed as estimates that require prudent interpretation, particularly with respect to streamflow in any specific year. In all of the simulations illustrated in this paper, an average and constant value for precipitation was used to produce the estimated water yield increases. In reality, precipitation is highly variable from year, as is streamflow.

The main reason we did not vary precipitation from year to year is that the WRENSS procedure produces estimates of evapotranspiration that are relatively insensitive to variations in precipitation until very low values of seasonal precipitation occur, i.e., the supply of available soil water begins to limit seasonal evapotranspiration. The WRENSS estimates of the water yield increases associated with a particular level of clearcut harvest tend to be fairly constant for a given set of climatic conditions over a wide range of precipitation. Since there is no provision in WRENSS or WRNSSDR to account for watershed storage changes from year to year, actual water yield increases will be much more variable than the simulated water yield increases.

Apportionment of Annual Increases to Daily Streamflow

We do not know of an easy or simple technique for evaluating the effects of harvest on daily streamflow. The WRENSS handbook (United States Environmental Protection Agency 1980) suggests a partial solution of apportioning the annual water yield to 7-day periods of a hydrograph from the area under investigation. There is certainly no evidence from Alberta experimental watershed results (Swanson et al. 1986) that forest harvesting speeds up the delivery of snowmelt water as was found at the Fool Creek watershed in Colorado (Leaf 1975), and suggested in the WRENSS handbook as a template for distributing the increases throughout the year. The composite hydrograph (Fig. 5) from the 18-catchment study conducted at Hinton (Swanson and Hillman 1977) is probably more typical of the distribution of the increased yield to events throughout the year, than that of the Fool Creek hydrograph in Colorado (Leaf 1975). A regional flood frequency analysis could provide the recurrence intervals for events of various magnitude, with a hydrograph similar to Figure 5, to provide an estimate of the potential change in the magnitude of snowmelt and rain streamflow peaks. Unfortunately, hydrographs describing the change in streamflow regime and volume following clearcutting are non-existent for most of the FMA areas of Alberta, the exception being the Hinton area (Fig. 5).

Hydrology of Mixed Species Stands

We do not have sufficient data on the hydrologic behavior of mixed coniferous–deciduous stands. The Boreal forest has extensive areas where conifers and deciduous trees share the same sites. We can only simulate what happens when all of the trees are harvested and a clearcut is regenerated with one species. The Boreal forest is an area where at least one clearcutting study would be invaluable to provide information on water yield changes and the hydrologic recovery period of such stands.

Watershed Storage in the Boreal Forest

We know too little about the role of watershed storage on dampening hydrologic extremes. Many watersheds in the Boreal forest contain large proportions of muskeg, beaver dams, and other wet sites. Topography may not be favorable for well-defined drainage patterns to be developed. Annual average water yields from Boreal watersheds in Alberta are generally less than 100 mm (Environment Canada 1991) indicating that precipitation is not much greater than evapotranspiration. Does forest harvesting have the same effects under these conditions as has been observed on more southerly watersheds where there is more precipitation and which respond relatively rapidly to precipitation events? In the absence of data to the contrary we must assume that they do, but considering the rapid regrowth of deciduous vegetation, it is possible that, given a sequence of low precipitation years, any increase in yield caused by harvest will fill vacant watershed storage and eventually be used by the regenerating or residual stand and never appear as streamflow.

Hydrologic Regimes for Fish Management

What sort of hydrologic regime is desired for proper fish management? Do fisheries managers want stable streamflow that varies little from year to year, or are periodic flushing events needed to maintain habitat? Forest managers cannot provide a

hydrologic stability greater than that resulting from the normal range of precipitation events, but they may be able to harvest in such a way that the water yield increases are reasonably stable over an extended period of time (Fig. 10), or occur as one large increase which declines rapidly (Fig. 9).

Using Available Techniques

We feel that existing techniques to assess land use impacts on aquatic resources are often ignored because they do not answer every hydrologic question. There are no easy or simple techniques to simulate daily streamflow or storm events from any watershed, much less those with little or no climate or streamflow data. The WRENSS procedure can assist the forester and fish manager by providing reasonable and objective estimates of the effects of forest harvesting on average annual water yield. The data-based WRNSSDR program, which incorporates regrowth, provides a relatively easy-to-use tool to estimate the cumulative effects of harvests through time as well.

There are no fixed formulae or even guidelines that we can apply in every instance that will guarantee that instream or downstream water users will be benefited or negatively impacted by forest harvest in a watershed. There has always been a lack of communication between water users and forest managers. At the present time, we have Alberta Environmental Protection's constraint of "15% of average annual water yield" as a sole criterion for water yield management anywhere in Alberta (John Taggart, Alberta Environmental Protection, Edmonton, Alberta, personal communication). An absolute criterion of how many millimetres of water yield that occur during specific downstream situations, such as flooding, would be desirable. We need similar criteria from fishery biologists for the flow volumes and flow regimes that they feel are desirable to maintain or improve fish habitat.

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Geologic Influences on the Response of Stream Channels to Timber Harvest-Related Impacts

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Abstract

We conducted watershed studies in northern California, U.S.A., on about 80 000 ha of redwood and Douglas-fir forest owned by the Pacific Lumber Company. The ownership was partitioned into five watershed assessment areas (WAAs) corresponding to the five major drainage basins within the study area. Extensive data on stream morphology, fish habitat, and riparian vegetation were analyzed and used to assess stream conditions in each WAA. Conditions among the WAAs differed, particularly with respect to sediment levels, riparian canopy, and pool size/abundance. We then examined the relationship between the observed habitat values and indicators of management intensity, such as hectares of recently harvested forest, and road density. Increased management activity was generally associated with reduced riparian canopy and increased water temperatures. However, most other indicators of habitat quality, including five substrate-related variables and three measures of pool size, showed either an inconsistent or a positive relationship to management intensity. To try to understand this paradox, we analyzed data on the parent geology and soils present in each WAA to determine if geologic differences might account for some of the variation in stream conditions. We found that WAAs with higher risk levels for mass wasting and surface erosion also tended to have poor habitat quality. Watershed assessment areas with a greater proportion of high-risk areas generally had higher fine sediment levels, lower gravel abundance, and smaller pool dimensions than did other, more stable WAAs. Our results suggest that local geologic factors may be more important than simple measures of management activity such as road density or hectares harvested in determining the effects of logging-related activities on streams.

Barrett, J.C., Alden, H.G., and Scholz, J.G. 1998. Geologic influences on the response of stream channels to timber-harvest related impacts. Pages 95–107 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

Watershed analysis efforts are currently being conducted throughout the Pacific Northwest. Most of these analyses attempt to identify how road construction and timber harvest affect the condition of stream channels and fish habitat. These studies report similar types of impacts: increases in sediment loading and transport, habitat simplification, modifications in discharge, and changes in channel morphology. However, the magnitude of these changes and their resultant effects on anadromous fishes vary widely.

Forest management can increase sediment delivery to streams via mass wasting, surface erosion of upland areas, and stream bank erosion (see review in Chamberlin et al. 1991). Road density, road placement and construction, yarding technology, and the intensity of harvest activity have all been implicated as factors that can determine the magnitude of sediment inputs (Burns 1972; Chamberlin et al. 1991; Washington Department of Natural Resources 1994). These sediment inputs can affect fish populations in several ways. First, they can result in the filling of deepwater habitats. The cover, slow current velocities, and, in some cases, thermal refugia provided by these deepwater habitats are important in determining fish abundance and production (Holtby 1987; Ozaki 1988; Matthews et al. 1994; Reeves et al. 1990). Second, fine sediments can clog the pores or interstitial spaces between individual sediment particles, reducing invertebrate production and egg/alevin survival of fish species that bury their eggs (Hall and Lantz 1969; Burns 1972; Erman and Mahoney 1983; Chapman 1988). Finally, at extreme loading rates, sediment inputs can result in changes in the entire structure or morphology of channels. Typical changes include decreased sinuosity and depth, and increased channel width. These alterations in channel form both reduce the abundance of quality fish habitat and allow for increased thermal loading to streams.

Timber harvest in riparian areas can make streams more susceptible to the negative effects of sediment inputs. Riparian harvest results in a decrease in large woody debris (LWD) recruitment to streams (Grette 1985; Andrus et al. 1988). Because LWD in streams often results in the storage of sediment and dissipation of stream energy, its loss can result in significantly increased transport of sediments to low gradient reaches used as spawning and rearing habitat by many fish species (Keller and Swanson 1979; Bisson et al. 1987; Swanston 1991). In addition, riparian harvest can also lead to decreased bank stability, making channels more likely to become wide and straight when sediment inputs increase.

We have conducted several watershed analyses to identify how timber harvest affects aquatic resources. Like many authors, we found considerable variation in the impact of timber harvest activities on both stream conditions and fisheries resources. For example, within a given geographic area logging roads might be responsible for 80% of all sediment production in one basin, and only 30–40% in another. Alternately, the effect of a given quantity of delivered sediment on fish varies, depending on the gradient and confinement of the stream channels being analyzed. This variation makes it difficult to develop management mitigations for sediment that are broadly applicable to a variety of sites, forest types, and silvicultural approaches.

A different but related problem is the difficulty in determining the degree to which habitat conditions are the current result of past management activities. The literature certainly provides abundant evidence of the negative impacts of logging and road construction on aquatic resources (e.g., Erman et al. 1977; Cederholm et al. 1981; Bisson et al. 1987). However, we have observed that logging activities are often automatically assumed to be responsible for habitat conditions that may have more to do with natural effects imposed by channel morphology, parent geology, and other physical aspects of the watershed than on past management. For example, pool frequency can be dictated as much or more by the density of flow obstructions (e.g., bedrock, outcrops, or boulders), as by the inputs of sediment from roads or harvest areas (Washington Department of Natural Resources 1994). Similarly, substrate characteristics, such as fine sediment and gravel levels, are determined in large part by the parent geology of a basin, the frequency and magnitude of scouring events, and channel confinement and slope. Ideally, data on conditions in unmanaged basins could be compared to data from managed forests to determine the net effect of logging activities. Unfortunately, unmanaged reference sites are not available for many locations and forest types. Data for larger streams, which more typically contain fish, are particularly scarce.

The problem of determining the net impact of logging has affected our most recent watershed analysis, an assessment of 77 000 ha of redwood forest owned by the Pacific Lumber Company (PL) along the North Coast of California. This area has natural rates of sediment inputs to streams that are

among the highest levels observed in North America (Kelsey 1980; Swanston 1991; Best et al. 1995). Still, because of extensive data showing that timber harvest activities lead to mass wasting and surface erosion, we had expected to find correlations between aquatic habitat conditions, particularly those related to sediment, and the past management history of basins in the study area. To our surprise, our initial work indicated that many of these relationships were weak, or, in some cases, exactly the opposite of what we would expect. In an attempt to understand this, we conducted a more extensive analysis of current conditions in the study area. We analyzed data on aquatic habitat variables, road density, harvest history, parent geology, soils, and slopes present in different portions of PL's ownership. This paper details our analysis of current conditions, as well as attempts to relate habitat variables to management history, and the mass wasting and surface erosion hazards within the watersheds we studied.

Methods

Study Sites

All studies were conducted on watersheds wholly or partly owned by the Pacific Lumber Company of Scotia, California. Pacific Lumber Company owns approximately 83 781 ha (206 940 acres) of land along the North Coast near the city of Eureka, California (Fig. 1). Overall, the forests are dominated by coast redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudotsuga menziesii*), although some hardwoods, including red alder (*Alnus rubra*), bigleaf maple (*Acer macrophyllum*), Oregon ash (*Fraxinus latifolia*), California laurel (*Umbellularia californica*), and various willow species (*Salix* spp.) are located in riparian areas. Dryer sites also contain tan oak (*Lithocarpus densiflorus*), manzanita (*Arcto-staphylos* spp.), and Pacific madrona (*Arbutus menziesii*). Off-ownership portions of the study area include both the forest types noted above, and grassland areas dominated by annual species. Annual rainfall of 125–250 cm occurs primarily from October to April (Kelsey 1980). Summer fog conditions are common, and fog drip from redwood trees is an important source of water during the dry months.

The ownership contains almost 200 named streams and rivers. These water bodies collectively contain a variety of fish species. Anadromous species include chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*O. kisutch*), steelhead trout (O. mykiss), sea-run cutthroat trout (*O. clarki*), and Pacific lamprey (*Lampetra tridentata*). Resident fish species include

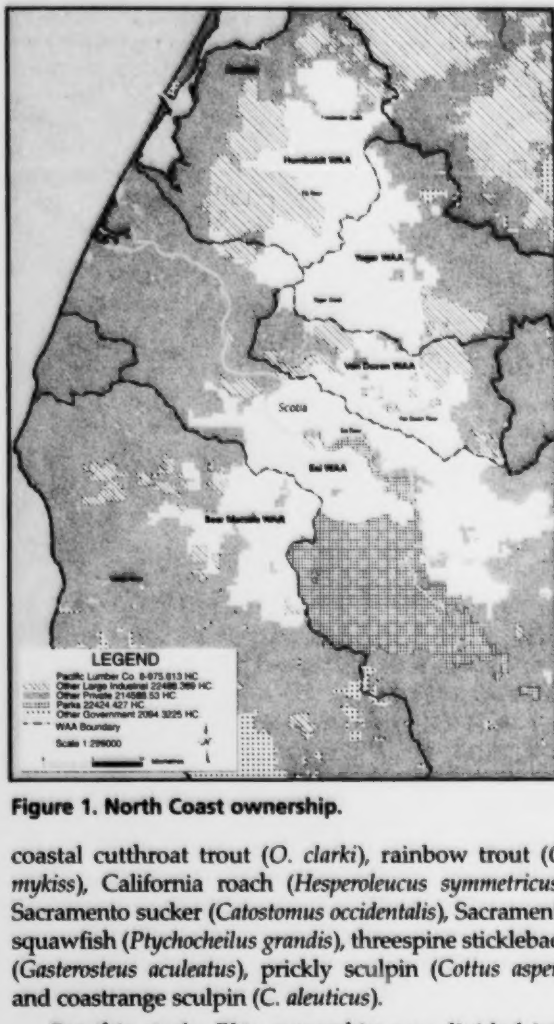

Figure 1. North Coast ownership.

coastal cutthroat trout (*O. clarki*), rainbow trout (*O. mykiss*), California roach (*Hesperoleucus symmetricus*), Sacramento sucker (*Catostomus occidentalis*), Sacramento squawfish (*Ptychocheilus grandis*), threespine stickleback (*Gasterosteus aculeatus*), prickly sculpin (*Cottus asper*), and coastrange sculpin (*C. aleuticus*).

For this study, PL's ownership was divided into five watershed assessment areas (WAAs): the Humboldt Bay WAA, Yager WAA, Van Duzen WAA, Eel WAA, and the Bear-Mattole WAA (Fig. 1). The Yager, Van Duzen, Eel, and Bear-Mattole WAAs contain all or part of the Yager Creek, and Van Duzen, Eel, and Mattole River watersheds, respectively. The Humboldt Bay WAA contains three major drainages—the Freshwater and Salmon creeks, and the Elk River—that all drain into Humboldt Bay. Analyses within this paper focus on overall conditions in each of these WAAs, rather than on site-specific conditions within subbasins or individual streams.

Four of the five WAAs are similar with respect to climate, vegetation, and topography. The Bear-Mattole WAA, however, is unique in that it is dominated by hardwood species, and Douglas-fir rather

Table 1. Characteristics of each watershed assessment area (WAA)

	Size (ha)				Vegetation type (%)						Slopes (%)		
WAA	PL	Non-PL	Major drainage	Road density (km/km^2)	Open/ young	Mid-successional	Late seral	Other[a]	Geology[b]	Soil types[c]	<30	30–0	>50
Humboldt Bay	14 848	52 088	Freshwater Creek, Elk River	2.56	24	24	42	10	FS,FM, TR	La, Hu, At, Me	49	41	10
Yager	13 951	34 255	Yager Creek	3.44	57	11	20	12	YF, FM, FS	Hu, La, At, Lg, Me, He	50	40	10
Van Duzen	10 149	22 419	Van Duzen River	2.76	32	34	22	12	YF, FM, FS	La, Hu, He, Me	38	40	22
Eel	30 852	180 167	Eel River	2.71	26	20	37	17	FS, YF	La, Hu, He, Me, At, Kn, Lg	30	38	32
Bear-Mattole	13 981	64 489	Bear and Mattole rivers	1.78	13	8	19	60	FS, FM, TR	Hu, Me, Kn, Wi, Lg, He	14	33	53

[a] Primarily hardwoods (alder, maple, manzanita, and tan oak) and grasslands.

[b] FM = Franciscan Mélange; FS = Franciscan Sandstone; TR = Tertiary Rock (Wildcat units); YF = Yager Formation.

[c] At = Atwell; He = Hely; Hu = Hugo; Kn = Kneeland; La = Larabee; Lg = Laughlin; Me = Melbourne; Wi = Wilder.

than redwood. Differences in stream conditions between this and the other WAAs, as noted below, could therefore be partly due to vegetation.

The WAAs differ in their management history. All of the WAAs have experienced some road building and timber harvest (Table 1), although portions of PL's ownership are unique among private landholdings on the North Coast in that they contain large areas with mature or late seral redwood forest. Many portions of the Humboldt Bay WAA, in particular, have experienced little timber harvest or road building since logging operations at the turn of the century. Over 42% of the land in this WAA is covered by late seral redwood forest, including several thousand acres of old growth stands. By contrast, the the Yager WAA has experienced intensive harvesting and road building over the last 15 years. Over 57% of the area in the Yager WAA either contains no forest (i.e., was very recently harvested) or young forest less than 20 years old. Road density shows a similar if less dramatic variation, ranging from a low of 1.78 km/km^2 in the Bear-Mattole WAA to 3.44 km/km^2 in the Yager WAA (Table 1). In general, the WAAs can be ranked with respect to the level of management activity as follows: Yager > Van Duzen > Eel > Humboldt Bay > Bear-Mattole.

The WAAs are located in the Coast Range of northern California. This area has experienced recent (post-Miocene) uplift of marine sediments, resulting in highly deformed and faulted sandstone, and mélange units of the Franciscan assemblage (Kelsey 1980). Specific geologic units include the Yager Formation, Franciscan Sandstone, Franciscan Mélange, and undifferentiated Tertiary rocks of the Wildcat group (Fig. 2). All of these units are unstable and highly erosive. The more competent sandstones and siltstones of the Yager Formation, Franciscan Sandstone, and Wildcat units are locally sheared, and characteristically contain moderate–steep, smooth-benched slopes, sharp ridge crests, and V-shaped canyons (Kelsey 1980; Winzler and Kelly 1980). Franciscan Mélange consists of highly sheared sandstone and interbedded siltstone, with inclusions of large igneous and metamorphic rocks. Mélange landscapes are rolling and hummocky, with highly unstable areas of earth flows and slumps (Winzler and Kelly 1980).

Overall, soils in the WAAs are dominated by the Hugo and Larabee soil series (Fig. 3), which are derived from each of the geologic units except

Figure 2. Geologic units on Pacific Lumber Company's ownership.

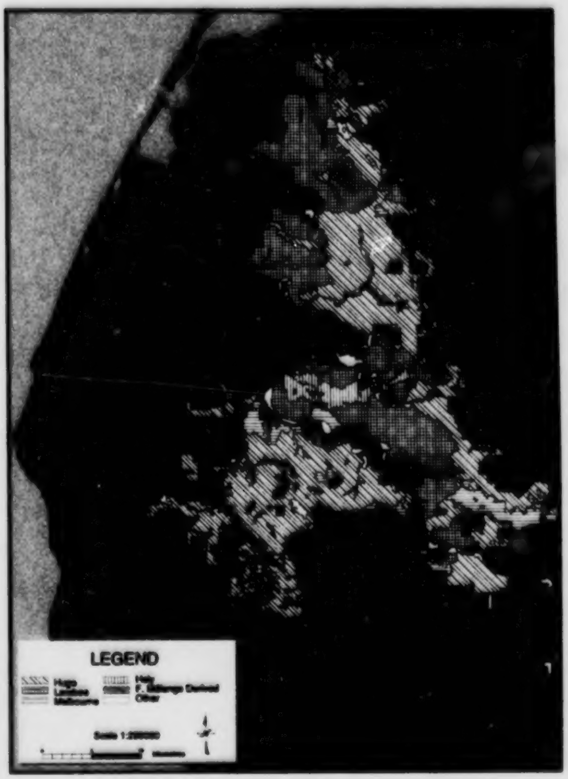

Figure 3. Soil types present on Pacific Lumber Company's ownership.

Franciscan Mélange. Hugo was the most common forest soil series observed. Hugo soil is typically well drained, loamy to clay soil from 75 to 152 cm deep (Winzler and Kelly 1980), with moderate stability and erodibility compared to other soils in this region. It was the most common soil series in the Yager and Bear-Mattole WAAs. Larabee series soils are similar to Hugo soils, but are finer textured and somewhat less stable. The Humboldt Bay, Van Duzen, and Eel WAAs contained large areas with Larabee soils. Hely and Melbourne soils are interspersed with the Hugo and Larabee series. Hely soils have stability and erodibility characteristics comparable to Hugo soils, whereas Melbourne soils are similar to Larabee soils. Franciscan Mélange weathers to Atwell and Kinman soils, which are notable for being highly unstable, and to Kneeland, Laughlin, and Wilder soils, which are similar in stability and erodibility to the Hugo and Larabee soils. Franciscan Mélange-derived soils were most evident in the Humboldt Bay and Bear-Mattole WAAs.

Data Sources

A variety of existing data sources were used to identify conditions in each WAA. These existing sources provided data on stream conditions, fish distribution and abundance, road density and type, harvest history, and vegetational characteristics on the ownership. Stream habitat inventory data were obtained from the California Department of Fish and Game's (CDF&G) geographic information system (GIS) database. These data were collected from 1989 to 1995 during streamside surveys by state personnel using methods in Flosi and Reynolds (1994). All measurements in this database are for individual habitat units (e.g., pools, riffles) that were sampled sequentially during pedestrian habitat surveys. Data for 53 streams on PL's ownership were used for our analysis.

Both the CDF&G and PL have conducted monitoring studies within streams on PL's ownership. California Department of Fish and Game's data on temperature and fine sediments in spawning gravels (i.e., fine sediments less than 0.84 mm in diameter)

were available for selected dates and times from 1991 to 1994. Pacific Lumber Company's monitoring data for fine sediments and temperature were available from 1994 and 1995.

Data on slopes, soils, stream locations and classes, and vegetation came from a variety of sources. Data on the geology and soils of the study area were obtained from the Natural Resource Conservation Service (i.e., Soil Conservation Service) and the California Department of Mines and Geology. Generally, geologic data were available for the on-ownership portions of each WAA and adjacent non-ownership lands. Data were not available for portions of the Humboldt Bay and Yager WAAs within PL's ownership. Slopes were determined with GIS using digital elevation models provided by the United States Geological Survey. The GIS was also used to identify road density, stream density, and watershed areas using data previously compiled by PL. Vegetation coverage was determined using aerial photographs, Spot™ satellite photography, and ground plots.

Data Analysis

Data collected for individual habitat units (e.g., individual pools and riffles) were initially aggregated by stream. However, many of the streams surveyed by the State included more than one channel type. Channel typing was based on Rosgen's classification system (Rosgen 1994), which considers entrenchment, width/depth ratios, sinuosity, slope, and substrate type. Because channel type, in part, determines the kind and abundance of fish habitat that can exist in a stream (Washington Department of Natural Resources 1994), these changes in channel type were potentially important. Consequently, data on habitat conditions were also aggregated into one of four channel types within each stream: cascading (Rosgen type A), riffle-dominated (Rosgen type B), meandering (Rosgen type C), and other (Rosgen types E, F). This aggregation provided data for 140 stream segments on PL's ownership.

Twelve habitat variables were examined: embeddedness, silt/sand dominance (defined below), fine sediments <0.84 mm, gravel dominance (defined below), sediment D_{50} (i.e., median sediment particle diameter), pool abundance, average pool depth, maximum pool depth, residual pool volume (i.e., pool volume at zero flow), in-stream cover, canopy cover, and maximum weekly average temperature (MWAT). Results presented here are the average values for all stream segments in each WAA. Ranks of the average values for each variable were also assigned by WAA, with a rank of 1 corresponding to the best conditions observed, and 5 corresponding to the worst conditions. Best and worst were judged using the habitat requirements of salmonids as a guide.

Except as noted below, average values for habitat variables were calculated using data from 15 streams and 35 stream segments in the Humboldt Bay WAA, 14 streams and 47 stream segments in the Yager WAA, 5 streams and 12 stream segments in the Van Duzen WAA, 21 streams and 37 stream segments in the Eel WAA, and 4 streams and 9 stream segments in the Bear-Mattole WAA. Data for the stream segments, in turn, include measurements from up to several hundred individual habitat units (e.g., pools, riffles, etc.).

Habitat data for fine sediments <0.84 mm, D_{50}, and MWAT come from a more limited number of monitoring sites maintained by the CDF&G and PL. Pacific Lumber Company has a network of 52 permanent monitoring stations distributed across its ownership. Data included here were obtained from 46 sediment monitoring sites, and 28 temperature monitoring stations. Data from 12 CDF&G permanent monitoring stations on PL's property were also used for fine sediments <0.84 mm and temperature, as available.

Stream habitat data for the variables of fine sediments <0.84 mm, sediment D_{50}, pool abundance, average pool depth, maximum pool depth, residual pool volume, and canopy underwent a quality assurance/quality control process but were otherwise used exactly as reported by the State of California and PL. Embeddedness, silt and sand substrate dominance, gravel substrate dominance, in-stream cover, and MWAT involved special analyses as follows:

- Embeddedness: This variable was estimated visually by the CDF&G, who recorded the following four ranges of embeddedness: 1) 0–25%; 2) 25–50%; 3) 50–75%; and 4) 75–100%. Thus, embeddedness scores are presented on a scale of 1–4, rather than the 0–100% scale typically used in other studies. Embeddedness scores of 1 and 4 were assumed equivalent to embeddedness levels of 12.5 and 87.5%, respectively. Scores from 1 to 2, 2 to 3, and 3 to 4 equaled embeddedness levels of 12.5–37.5%, 37.5–62.5%, and 62.5–87.5%, respectively.
- Silt and Sand Dominance: The CDF&G did not include measurements of the percentage

of the stream bottom composed of each of several size classes of sediment. Instead, the CDF&G recorded the most abundant particle size class (e.g., sand, cobble) present at each pool tailout encountered (Flosi and Reynolds 1994). These data were used to calculate the percentage of all pool tailouts that were dominated by silt or sand substrates.

- Gravel Dominance: Similar to silt and sand dominance, this metric measured the proportion of pool tailouts that were dominated by gravel. For the salmonid fishes found on the ownership, high gravel dominance at pool tailouts is a positive condition. For all other sediment measurements, embeddedness, percent fines, and silt/sand dominance, high values are associated with negative conditions for salmonids.
- In-stream Cover: The CDF&G measured in-stream cover in a two-step process. First, the percentage of the active channel containing cover for fish was visually estimated. Second, the value of this cover was rated subjectively, then used to weight the estimate of total cover present. We used only data from step 1—estimates of total cover present in each habitat unit.
- MWAT: Water temperature (°C) was measured by continuously recording thermographs. These continuous temperature readings were used to calculate the maximum weekly average temperature, that is, the highest average temperature observed during any consecutive 7-day period. Most thermographs recorded temperature hourly, so our averages were usually based on 168 consecutive measurements.

Data on slopes and parent geology were used to develop an index of mass wasting risk for lands within each WAA. Previously, PL and Pacific Watershed Associates of Eureka, California had analyzed mass wasting risk using a four-factored index that included two measures of geologic conditions: slope and mass wasting hazard maps developed by the California Department of Mines and Geology. In the analysis presented here, we used two of the factors included in the previous index—parent geology and slope—to develop a second index of mass wasting hazard. With our approach each parent geology type present on the ownership received a score of 2 to 4 depending upon its inherent stability. Slopes were accorded scores of 1 to 12 depending upon gradient and position. Slope scores increased with increasing gradient, and were higher for inner gorge sites, defined here as slopes of 50% or more that led, uninterrupted, to a stream channel. Scores for geologic type and slope were then combined, and the resulting values assigned to one of four categories: low, moderate, high, or extreme. Low-risk ratings applied to all areas with slopes less than 35%, and to slopes from 35 to 50% when the most stable parent geologic types were present. Moderate ratings excluded all inner gorges or slopes greater than 65%, but included slopes from 35 to 65% in the second most stable group of geologic types, and slopes of 35 to 50% for the least stable types. High ratings included slopes of 50% or greater containing the least stable parent geologic types, and slopes greater than 65%, or from 50 to 65% if inner gorges were present, for the two most stable geologic types. Extreme ratings applied to inner gorge slopes containing the least stable geologic types, and all inner gorge slopes with a gradient exceeding 65%.

Surface soil erosion hazard ratings (EHRs) were determined using the California State Board of Forestry Technical Rule Addendum Number 1 (State of California 1997). This measure is widely used in California to assess whether roads and harvest areas are likely to experience significant surface erosion. The rating incorporates several erosion-related factors including: 1) soil characteristics, 2) slope, 3) vegetative cover, and 4) precipitation. As with our mass wasting hazard index, each factor is assigned a score. These scores are then summed, and the sum used to determine the EHR using standards for low, moderate, high, and extreme risk set by the State.

Results

Sediment Variables

Sediment variables included embeddedness, silt/sand dominance, fine sediments <0.84 mm, gravel dominance, and sediment D_{50}. Embeddedness values in all WAAs had average values between 2 and 3 (Table 2), corresponding to embeddedness levels of 37.5 to 62.5%. Embeddedness values were highest in the Eel and Bear-Mattole WAAs, lowest in the Yager WAA, and intermediate in the other two study areas.

The percent of all pool tailouts having silt and sand as the dominant substrate varied from 10.6 to 25.9% among WAAs (Table 2). The Yager WAA had the lowest occurrence of silt/sand substrates, and the Humboldt Bay WAA had the highest. Gravel substrates were more commonly found in pool tailouts, representing the dominant substrate in 33 to 45% of all observations (Table 2). Sand/silt abundance and gravel abundance tended to be inversely related. The two WAAs with the highest silt/sand concentrations, Humboldt Bay and Bear-Mattole, also had the lowest gravel dominance values. Conversely, the Yager WAA had both the lowest silt/sand dominance and the highest gravel dominance.

Substrate samples collected from each WAA contained an average of 16.6 to 29% fine sediments <0.84 mm in diameter (Table 2). Predictably, Yager WAA, which had the lowest silt/sand dominance, also had the lowest fine sediment concentrations. The Humboldt Bay WAA showed a similar but opposite pattern of having high silt/sand dominance and high fine sediment levels. Average fine sediment levels exceeded 20% in three of the five WAAs.

The average D_{50} values varied from 57 to 93 mm among the WAAs (Table 2). Values for this variable were high in the Eel and Yager WAAs and relatively low in the other WAAs.

Collectively, the results for the sediment variables indicate that the Humboldt Bay and Bear-Mattole WAAs had high levels of fine sediment relative to levels in the other study areas. By contrast, the Yager WAA consistently had the best values for sediment, ranking number one in each of the variables examined. The Van Duzen and Eel WAA were less consistent, alternating between fair and poor conditions with respect to sediment, depending upon the variable being examined.

Table 2. Average values for stream habitat conditions in the watershed assessment areas (WAA). Values within parentheses represent the rank with 1 being the best observed and 5 the worst.

Stream habitat variable	Humboldt Bay WAA	Yager WAA	Van Duzen WAA	Eel WAA	Bear-Mattole WAA
Embeddedness score	2.6 (3)	2.5 (1)	2.5 (2)	2.7 (4)	2.9 (5)
Silt and sand substrate dominance (%)	25.9 (5)	10.6 (1)	15.8 (2)	16.4 (3)	22.3 (4)
Fine sediments (<0.84 mm) (%)	26.6 (4)	16.6 (1)	29.0 (5)	23.8 (3)	18.3 (2)
Gravel dominance (%)	35.0 (4)	45.0 (1)	38.0 (3)	41.0 (2)	33.0 (5)
Sediment D_{50} (mm)	57.0 (5)	93.0 (1)	60.0 (4)	88.0 (2)	64.0 (3)
Pools (%)	45.0 (1)	22.0 (3)	14.0 (5)	23.0 (2)	15.0 (4)
Average pool depth (m)	0.26 (5)	0.51 (1)	0.37 (3)	0.36 (4)	0.39 (2)
Maximum pool depth (m)	0.64 (5)	0.90 (1)	0.70 (3)	0.68 (4)	0.71 (2)
Residual pool volume (m^3)	10.20 (5)	51.26 (1)	14.47 (3)	24.8 (2)	10.28 (4)
Cover (%)	17.0 (3)	20.0 (1)	17.0 (3)	17.0 (3)	11.0 (5)
Canopy (%)	76.1 (1)	50.0 (4)	68.0 (2)	54.0 (3)	15.0 (5)
Maximum weekly average temperatures (°C)	15.6 (2)	16.1 (3)	15.4 (1)	17.1 (4)	18.3 (5)

Pool Variables

Pool variables included pool frequency, average pool depth, maximum pool depth, and residual pool volume. Several patterns were evident in the data for these variables. First, streams in the Humboldt Bay WAA had a much greater percentage of their length composed of pools than did streams in any other WAAs (45%), but these were the shallowest and smallest pools observed anywhere on the ownership (Table 2). Second, the Yager WAA had deeper and much larger pools than did any other WAA. Third, average pool abundance was low in all areas except the Humboldt Bay WAA, ranging from 14 to 23% in the other assessment areas. Finally, pool depth values in the Van Duzen, Eel, and Bear-Mattole WAAs were similar, ranging from 0.36 to 0.39 m for average depth, and 0.68 to 0.70 m for maximum depth.

Other Habitat Variables

In-stream cover levels were low in all WAAs, ranging from 11 to 20%. Three of the five WAAs (Humboldt Bay, Van Duzen, and Eel) had the same average cover level of 17%, and a fourth (Yager) had nearly the same level (Table 2). Only the Bear-Mattole WAA, at 11%, had in-stream cover levels that differed markedly from those observed in other parts of the ownership.

As might be expected, values for canopy cover and water temperature (MWAT) appeared to be related. The two WAAs with the highest canopy levels, Humboldt Bay and Van Duzen, also had the lowest MWAT values. Conversely, the Bear-Mattole WAA had the lowest canopy cover and the highest MWAT values. Canopy levels were notable for the variation seen in average levels among WAAs, ranging from a high of 76.1% to a low of 15% (Table 2). Maximum weekly average temperature values were more restrained, with only 2.9°C separating the highest and lowest averages.

Slope Stability and Erosion

Over half of all lands within the Humboldt Bay, Yager, Van Duzen, and Eel WAAs were calculated to have a low risk of mass wasting, with estimates ranging from 50.5 to 60.5% of the WAA area (Table 3). By contrast, only 28.3% of the Bear-Mattole WAA consisted of low-risk areas. Moderate-risk areas were the next most abundant category, with 28.7 to 33.2% of all WAA lands falling within this category. Collectively, lands with low and moderate risk of mass wasting represented approximately 80 to 90% of the Humboldt Bay, Yager, Van Duzen, and Eel WAAs, but only 59% of the Bear-Mattole WAA.

Table 3. Mass wasting and surface erosion risk ratings for each watershed assessment areas (WAA). Values represent the percentage of each WAA having a given risk rating.

	Humboldt Bay WAA (%)	Yager WAA (%)	Van Duzen WAA (%)	Eel WAA (%)	Bear-Mattole WAA (%)
Mass wasting risk					
Low	58.04	57.60	60.53	50.49	28.29
Moderate	33.25	30.42	29.37	28.69	30.92
High	6.99	8.45	7.62	13.98	16.20
Extreme	1.72	3.53	2.47	6.84	24.58
Sum M–E	41.96	42.40	39.47	49.51	71.71
Sum H–E	8.71	11.99	10.10	20.82	40.79
Surface erosion					
Low	78.26	87.57	64.01	61.49	41.06
Moderate	21.37	12.28	33.31	36.12	53.81
High	0.01	0.06	0.63	0.48	4.64
Extreme	0.36	0.08	2.05	1.90	0.49
Sum M–E	21.74	12.43	35.99	38.51	58.94
Sum H–E	0.37	0.15	2.68	2.38	5.13

Our experience with PL's ownership indicates that areas with high and extreme risk for mass wasting and surface erosion are responsible for a majority of all sediment that is delivered to streams. Three of the four WAAs had similar areas in high- and extreme-risk categories for mass wasting: Humboldt Bay (8.7%), Yager (12%), and Van Duzen (10.1%). The Eel WAA, at 20.8%, contained approximately twice these levels, while the Bear-Mattole WAA contained about four times as much land in these higher-risk categories (40.8%). Based on these data, we would expect that the Eel and the Bear-Mattole WAAs, respectively, would have higher and much higher rates of sediment inputs to streams than occurs in the remaining study areas. However, even in the three more stable WAAs, high- and extreme-risk categories apply to approximately 10% of all lands, indicating a high potential for mass wasting-related sediment delivery to streams.

Our analysis of surface erosion was similar to the mass wasting evaluation in that the Humboldt Bay and Yager WAAs had low risk levels and the Bear-Mattole WAA had relatively high risk levels (Table 3). Low-erosion-risk areas covered 41.1 to 87.6% of all lands, depending on the WAA—a much greater range than was observed for mass wasting. However, low- and moderate-risk scores collectively applied to over 94% of all WAA lands. Thus, high- and extreme-erosion-risk areas were relatively limited, with three groups evident: a very low group consisting of the Humboldt Bay and Yager WAAs (0.4 and 0.2%, respectively), an intermediate group consisting of the Van Duzen and Eel WAAs (2.7 and 2.4%, respectively), and the upper value observed in the Bear-Mattole WAA (5.1%).

Discussion

Our watershed analysis of PL's lands had two goals: to summarize physical and biological information for the ownership and to assess the relative impact of various management practices used by PL. An extensive database on habitat conditions in area streams, and existing GIS coverages for roads, harvest history, slopes, soils, and parent geology simplified our analysis of existing conditions. We found that much of PL's ownership contains unstable geologic units, which, when combined with locally steep slopes, result in high and extreme mass wasting risk on 8.7 to 40.8% of the lands within each WAA. Surface erosion risk was less significant, but high- and extreme-risk areas still occupied 2 to 5% of most WAAs. Given the magnitude of these risk factors, we were not surprised to find that several variables sensitive to sediment inputs, including embeddedness, fine sediments <0.84 mm, and pool abundance, were at levels in all of our study areas that result in fair to poor habitat for salmonids. Results for other variables, however, showed more geographic variation. Observed values for silt/sand dominance, gravel dominance, sediment D_{50}, pool depths and volumes, canopy cover, and maximum weekly average temperature ranged from good to poor for salmonids, depending upon the WAA being examined.

We have had more difficulty with our second goal – assessing the impact of management practices by PL. Initially, we assumed that we would see relationships between habitat conditions and either specific management practices, such as building roads near streams, or overall measures of management intensity, such as road density or recent harvest levels. We expected to find that increased levels of management would be associated with poorer habitat conditions for fish. Once these habitat-management relationships were determined, we planned to use the results to identify specific management impacts, or, at least to determine threshold levels of management below which habitat conditions remain favorable. Pacific Lumber Company's lands are well suited to this type of analysis because extensive habitat data are available to assess current conditions, and because the ownership contains distinct hydrologic units that have very different levels of management.

Ultimately, the data did not support this approach to assessing management impacts. Instead, we found a poor correlation between management intensity and the majority of our habitat variables. The most striking example of this are the results for the Yager WAA. This area has been extensively roaded and harvested in the last 15 years. Of our five study areas, it currently has both the highest road density (3.44 km/km^2) and the greatest percentage of its surface area in recently harvested lands (57%). Indeed, the intensity of PL's management activities in the Yager WAA has been the focus of protests by environmental groups, and of regulatory concern by state and federal agencies. Yet this WAA had the best observed conditions for 9 of the 12 habitat variables examined, including all 5 of the direct measures of sediment condition, embeddedness, silt/sand dominance, fine sediments <0.84 mm, gravel dominance, and sediment D_{50}, and 3 of the 4-pool related variables: average pool depth, maximum pool depth, and residual pool volume.

Results for the Humboldt Bay and Bear-Mattole WAAs also demonstrate the poor correlation between the intensity of management and habitat variables.

Approximately 42% of the Humboldt Bay WAA consists of late seral forests, many of which have not been harvested in 100 years or more. The road density, at 2.56 km/km^2, was the second lowest observed, as was the percentage of the WAA composed of recently harvested areas (24%). Yet this area had the worst observed conditions for silt/sand dominance, sediment D_{50}, average pool depth, maximum pool depth, and residual pool volume, and the second worst conditions for fine sediments <0.84 mm and gravel dominance. The Bear-Mattole WAA had the lowest surface area in open or young forest, and the lowest road density of any of the WAAs. Yet it had the worst or second worst average values for embeddedness, gravel dominance, silt/sand dominance, pool abundance, and residual pool volume.

A possible explanation for this paradox is that differences in geologic conditions among the WAAs strongly influence stream habitat. The mechanism for this could be variation in natural or background inputs of sediments, a differential susceptibility to the effects of management, or both. Our analysis generally supports the hypothesis that higher levels of risk for mass wasting and surface erosion are associated with poorer habitat conditions. This is especially apparent for the Yager and Bear-Mattole WAAs. The Yager WAA had the lowest area in high-extreme surface erosion risk, and was only 3.3 percentage points higher than the lowest observed score for high-extreme mass wasting risk. Thus, the Yager WAA had comparatively low risk scores. As noted above, this WAA also contained the best habitat conditions for 9 of the 12 variables we examined. The Bear-Mattole WAA, by contrast, had the highest risk ratings for both surface erosion and mass wasting. These risk ratings were almost double the next highest values observed. Thus, we were not surprised to find that this WAA also had the worst or second worst observed habitat conditions for 8 of our 12 variables. Results for the Van Duzen and Eel WAAs also generally support a link between sediment risk factors and habitat conditions. These WAAs had intermediate mass wasting and surface erosion risk levels and also contained intermediate habitat values for most variables.

Results from the Humboldt Bay WAA, however, do not suggest a linkage between geologic factors and habitat conditions. The Humboldt Bay WAA had the lowest area in high-extreme mass wasting risk, and the second lowest observed score for surface erosion risk. Yet, this WAA had the worst or second worst observed values for 7 of the 12 habitat variables. The only unique geologic factor we could find to explain these results is the prevalence of Wildcat geologic units in the Humboldt Bay WAA (Fig. 2). It may be that our evaluations did not properly weight the sediment risk associated with these units. Alternately, impacts of past management activity, as discussed below, may be degrading habitat conditions more than would be expected based on geologic factors or current management activity alone. Regardless, this anomaly in our results indicates that more analysis of the relationship between geology and habitat conditions on PL's lands is needed before any causal link between them can be confirmed.

Our analysis of geologic factors focused on risk rather than specific mass wasting or erosion events known to have occurred in the WAAs. Several of the streams in our study area, particularly in the Eel and Bear-Mattole WAAs, have experienced catastrophic mass wasting events in the last decade associated with large storms. This suggests an alternate hypothesis. Given that the entire study area is unstable geologically, it may be that stochastic patterns of mass wasting determine much of the sediment loading observed at a landscape level. Under this hypothesis small, high intensity storms may hit portions of the landscape, creating sediment conditions that do not clearly link to management history, or possibly even attributes such as slope and vegetation. We are currently pursuing this hypothesis through an evaluation of the landslide history of each WAA.

We did note a weak link between management and two of the habitat variables we examined: canopy cover and water temperature. Canopy levels were highest in the Humboldt Bay WAA, intermediate in the Van Duzen and Eel WAAs, and second lowest in the Yager WAA. This ranking is similar to a ranking based on the percentage of each WAA composed of open land/young forest. With respect to temperature, the removal of streamside trees leads directly to increases in solar loading, which in turn leads to higher temperatures. Not surprisingly then, areas with low canopy cover also tended to have the highest temperature levels.

Our ability to link management history to canopy cover and temperature, albeit weakly, contrasts with the inconclusive or even counterintuitive relationship of management to the remaining habitat variables we examined. We do not doubt that management activities lead to sediment inputs to streams, or even that higher levels of management lead to higher sediment inputs. Pacific Lumber Company's management activities have clearly resulted in increased sediment delivery to ownership streams through

mass wasting of fill slopes from roads and landings, activation of slumps within harvest areas, and erosion of fine sediments from disturbed areas and unsurfaced roads. However, our data indicate that these increased sediment inputs do not necessarily translate into poorer habitat conditions for aquatic species. Instead, the relationship between management and habitat conditions in our study area appears to be more complex than that. Factors such as stream size, channel type, and management history prior to implementation of more stringent environmental requirements may all be important in determining the extent to which management activities impact habitat conditions for fish.

The effect of environmental regulations may be especially relevant here. The Humboldt Bay WAA was harvested from the 1880s to 1920s using environmentally destructive techniques including large clearcuts covering thousands of hectares, stream channelization, downhill yarding that dragged large quantities of soil to streams, and extensive post-harvest burning. In addition, many of the stream crossings from that period depended upon logging debris for water passage under road fill. These Humboldt crossings remain in many locations and, as they age, break down. Eventually they stop passing flows and fail, often delivering large quantities of sediment to streams in the process. Thus, even though harvest in this WAA occurred up to 120 years ago, it is possible that negative impacts from that harvest remain. By contrast, all recent harvest in the Yager WAA occurred under forest practice rules in California that, since 1973, have regulated road construction, and limited operations in and adjacent to streams, and on steep slopes. It seems possible that these rules allowed the Yager WAA to sustain greater management intensity with less environmental impact than would be expected given the effects of older harvests.

Although more work is needed to confirm and elaborate upon our results and analysis, this study suggests that interpreting the effects of PL's forest practices upon streams, and the selection of mitigation measures to prevent these effects, should be done with caution. In the absence of more definitive information there has been a tendency for regulating agencies and the public to assume that any problems with stream habitat or fisheries production on PL's lands are due to management activities. Our inability to show stronger links between management activity and stream conditions suggest that this assumption may not be correct for many habitat variables. Similarly, our hypothesis that local geologic factors exert a strong influence on habitat conditions indicates that state forest practice codes containing fixed requirements for road construction, yarding techniques, and harvest levels are probably going to be too conservative for some low-risk watersheds and too permissive in other, high-risk watersheds. Recognition of the limits of "one size fits all" approaches partly explains the recent increase in the use of watershed analysis techniques that focus on site-specific conditions and management impacts (e.g., Washington Department of Natural Resources 1994). These type of analyses permit a more customized approach to forest management whereby high-hazard portions of an ownership are subject to protective approaches aimed at maintaining stable conditions, and low-risk areas are subjected to more intensive management techniques. Pacific Lumber Company is currently working with state and federal regulators to develop this type of management approach as part of a habitat conservation plan being prepared for its lands.

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Stream Fertilization as a Fisheries Mitigation Technique for Perturbated Oligotrophic Trout Streams in British Columbia

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Abstract

Stream fertilization is a promising habitat restoration technique that is being supported within the Watershed Restoration Program of British Columbia, a program designed to restore watersheds impacted by past timber harvesting practices. In impacted systems, the addition of fertilizer replaces lost nutrients and increases food chain productivity, thereby improving conditions for fish to survive stressed conditions caused to the stream bank by activities such as logging. Three rivers are being experimentally treated: Adam River and Big Silver Creek, located on the south coast, and Mesilinka River, in the northern interior. The Mesilinka River is a large northern river (mean summer flow, 112 $m^3 \cdot s^{-1}$) located 280 km north of Prince George, and flows east into Williston Reservoir which empties into the Peace River. It is one of several oligotrophic streams inhabited by migratory and resident salmonids that were affected by construction of the reservoir, thereby flooding the lowermost and most productive stream reaches. To offset the lost productivity of fish habitat, liquid fertilizers (agriculture-grade nitrogen and phosphorus) were added to the river during the summer months of 1994 and 1995, after 2 years of pre- fertilization monitoring (1992 and 1993) in control and treatment reaches. Target in-river concentrations were very low, namely 5 $\mu g \cdot L^{-1}$ dissolved inorganic phosphorous and 20 $\mu g \cdot L^{-1}$ dissolved inorganic nitrogen.

Koning, C.W., Ashley, K.I., Slaney, P.A., and Paul, A.J. 1998. Stream fertilization as a fisheries mitigation technique for perturbated oligotrophic trout streams in British Columbia. Pages 109–120 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Preliminary results (1995) from 5–8 km index reaches suggest rainbow trout and mountain whitefish numbers have increased two-fold and five-fold, respectively. By 1995, weight-at-age of rainbow trout (age 4+ fish only) increased in the treated reach (T2) compared to the control (34%). The response of the food chain, indicated by periphyton accrual, was detectible for >15 km below the fertilizer drip stations. These, and earlier reported results demonstrating strong growth response of fish, indicate that low-level stream fertilization can also be used as one technique (usually within a larger set of habitat measures) in restoring impacted oligotrophic streams. New initiatives in our nutrient addition projects include development and application of a flow-proportional fertilizer injector system for liquid nutrients, and development and application of slow-release solid fertilizer briquettes for annual applications.

Introduction

Research has been conducted on stream fertilization for over a decade on Vancouver Island, British Columbia, first at the Keogh River during the 1980s (Slaney et al. 1986; Johnston et al. 1990), more recently at the Salmon River (Slaney and Ward 1993; Slaney, Ashley, Wightman, Ptolemy, and Zaldokas 1994), and also on a tundra stream in Alaska (Deegan and Peterson 1992). The primary objectives were to determine the effect of nutrient additions on the growth and abundance of anadromous salmonids in oligotrophic streams (Fig. 1), to determine if controlled seasonal enrichments are a cost-effective rehabilitation technique, and to evaluate the technique in order to provide mitigation of logging impacts on, for example, overwinter survival of juvenile and adult salmonids. Fertilization of the Keogh and Salmon rivers resulted in up to 2- to 3-fold increases in the average weight of juvenile steelhead trout after only 2 to 3 months of fertilization (Fig. 2) (Slaney et al. 1986; Johnston et al. 1990; Slaney and Ward 1993). Periphyton responses were detected 50 km downstream from a single fertilizer input site at the upper Nechako River (northern British Columbia), illustrating that nutrient cycling or spiralling can occur over extended distances in large streams (Slaney, Rublee, Perrin, and Goldberg 1994).

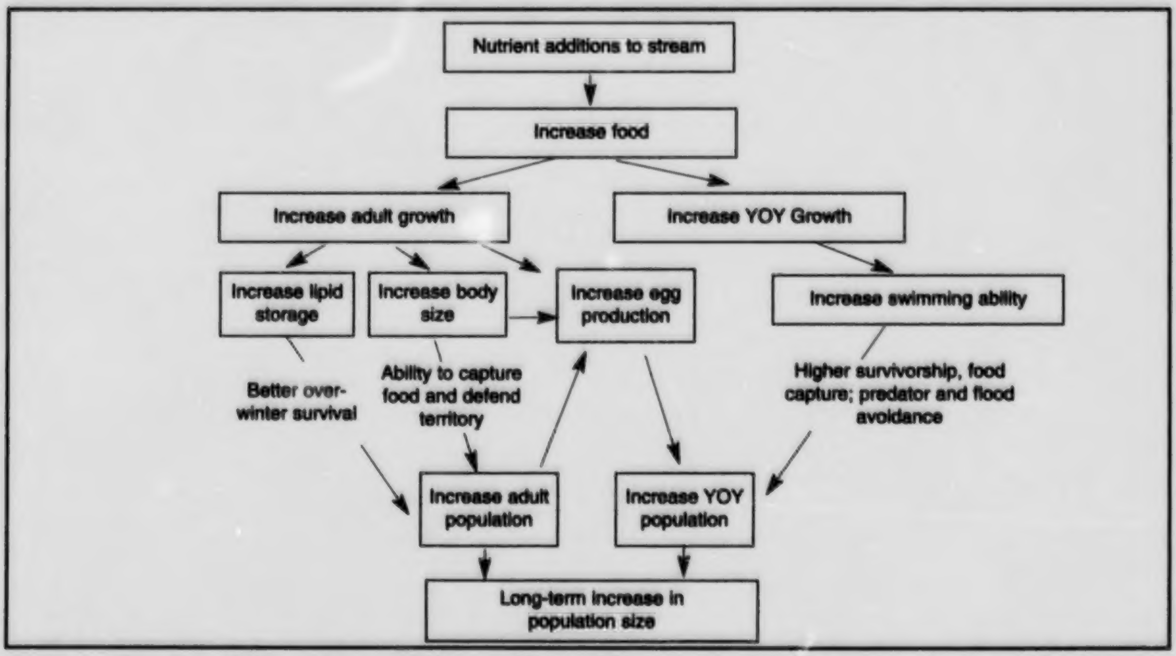

Figure 1. Overview of the response of fish to nutrient additions to the stream, based on Arctic grayling response (Deegan and Peterson 1992).

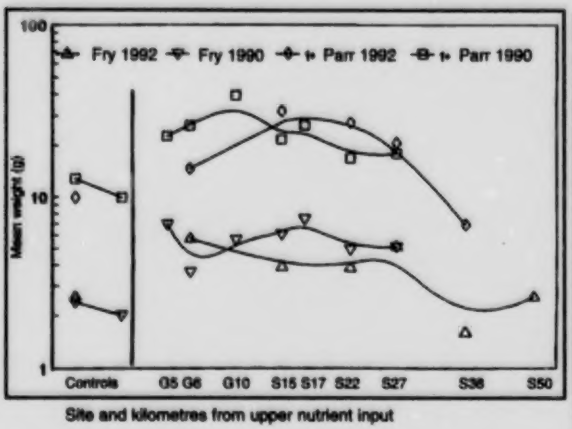

Figure 2. Trends in mean weights of rainbow (juveniles and adults) and steelhead trout (juveniles) within control, fertilized, and downstream sampling sites in Grilse Creek (which flows into the Salmon River) and the Salmon River during 1990 and 1992. Spatial controls were located in the upper Salmon River, Grilse Creek (G), and the mainstem of the Salmon River (S), with distances in km from the upper fertilizer tank at Grilse Creek (Slaney and Ward 1993).

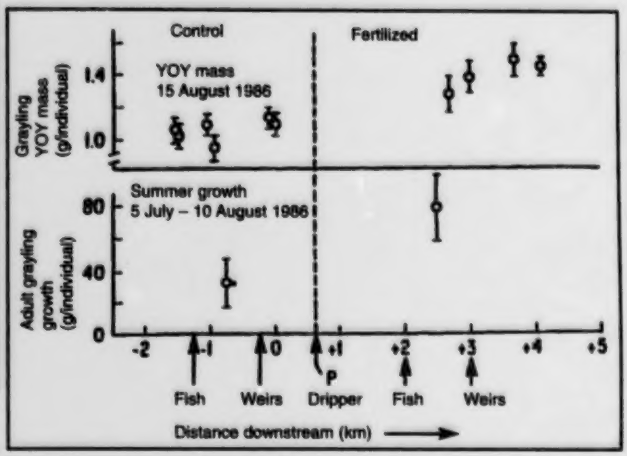

Figure 3. Arctic grayling growth in control and fertilized reaches in 1986. Upper: the mean mass of young-of-the-year (YOY) grayling was significantly larger at sites below the phosphorous addition site compared to sites above ($t = 9.7$, df - 294, $P < 0.001$). Lower: the mean mass gain of adult grayling in the fertilized reach during 5 weeks of July and August 1986 was significantly higher than mass gain in the control reach ($t = 2.6$, df - 39, $P = 0.01$) (Peterson et al. 1993).

Insect bioassays confirmed there are strong benthic insect responses to nutrient additions (Quamme 1994; Perrin and Richardson, in prep.).

A few trophy-sized resident trout fisheries have been historically associated with cultural enrichment, for example, the Cowichan River (Vancouver Island), and Crowsnest and Bow rivers (Alberta), suggesting that inorganic nutrients from treated sewage effluent can be beneficial if discharged to the river in a controlled manner to avoid contaminants. The response of resident salmonids is likely more evident than with anadromous fish because they remain in the stream for several more years, especially among species with some longevity. With only a six-week growth window in the Kuparuk River in northern Alaska, adult grayling in the fertilized reach gained an average of 78 g (20.6%), while fish in the control reach gained an average of 32 g (8.9%). Young-of-the-year grayling (age 0+ fish) were 40% heavier in the fertilized reach than in the control (Fig. 3) (Petersen et al. 1993).

The establishment of the Williston Reservoir in 1968–1972 flooded substantial portions of the Peace, Parsnip, and Finlay rivers, as well as the lower portion of many large and small tributaries, including the Mesilinka and Nation rivers (Barrett and Halsey 1985). Riverine habitats (such as groundwater channels, off-channel ponds, oxbows, and deep pools) utilized by salmonids, including Arctic grayling, and mountain whitefish were reduced substantially (Bruce and Starr 1985). Thus, the present Mesilinka fertilization project is a mitigation option, to compensate for loss of fish populations in the lower, flooded reaches. The purpose of this paper is to describe the initial results from this investigation, and to describe some of our new developments in stream fertilization work.

Study Area

The Mesilinka River is a large northern river (watershed area, 3285 km^2) located about 280 km north of the city of Prince George, B.C. The headwaters originate in the Omineca mountain range and the river flows for about 120 km before emptying into B.C.'s largest freshwater body, Williston Reservoir (Fig. 4). Williston Reservoir (watershed area, 70 860 km^2; reservoir surface area, 1775 km^2) was formed behind the W.A.C. Bennett Dam during the late 1960s to provide hydroelectric energy, and is part of the Peace–

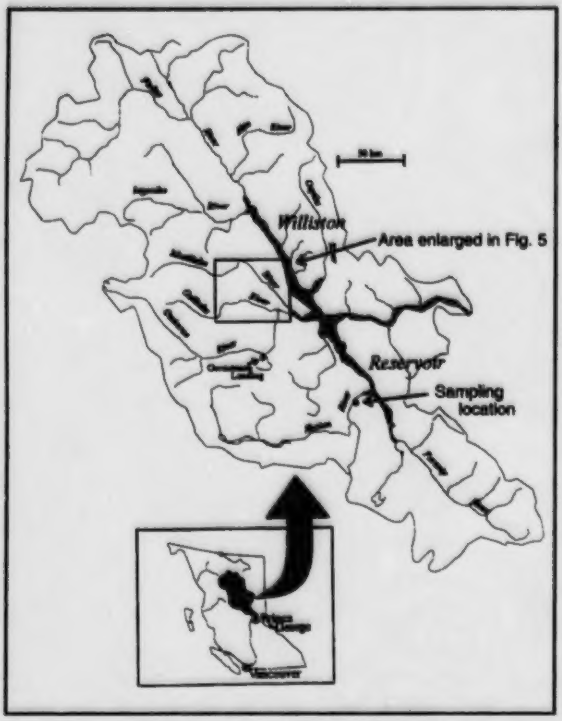

Figure 4. Project area, Mesilinka River located in northern British Columbia.

Slave–Mackenzie River system, which ultimately flows north and discharges into the Arctic Ocean.

Flows of the Mesilinka River are relatively high in spring to early summer due to melting snowpack and spring rains. Mean spring to summer flows for the 10-year period of 1982–91 were 74, 180, 108, 49, and 35 $m^3 \cdot s^{-1}$ during May, June, July, August, and September, respectively (Water Survey of Canada data). Minimum flows of 6 to 8 $m^3 \cdot s^{-1}$ occur in February and March. The temperature regime of the Mesilinka River is typical of many large Williston Reservoir streams, averaging 10–13°C (mean monthly) in summer.

Concentrations of soluble reactive phosphorous (SRP) and total dissolved phosphorous (TDP) are extremely low, typically below detectable limits of <1 $\mu g \cdot L^{-1}$ and <3 $\mu g \cdot L^{-1}$, respectively, indicating that the river and its main tributaries are highly oligotrophic. Nitrate–nitrogen ranges from 5 to as high as 40 $g \cdot L^{-1}$ during the summer season. Nitrate–nitrogen values below 20 $g \cdot L^{-1}$ are considered to be algal growth limiting (Slaney, Rublee, Perrin, and Goldberg 1994). Periphyton accrual is very low (peaking at 10–16 $mg \cdot m^{-2}$), which corresponds with low insect abundance in stream substrates. Underwater inspection of the substrate of several riffles in the main stem confirm that the periphyton and insect communities are poorly developed and of low biomass. The low nutrient concentrations are typical of a northern interior watershed of shallow soils overlying bedrock, and without returning salmon or kokanee to provide an external source of marine- or reservoir-derived nutrients.

Wild rainbow trout (*Oncorhynchus mykiss*), Arctic grayling (*Thymallus arcticus*), and mountain whitefish (*Prosopium williamsoni*) populations inhabit the river, as well as smaller populations of bull trout (*Salvelinus confluentus*). Aside from bull trout, adult salmonid fish are small in size (≤ 30 cm on average). Salmonid spawning and rearing habitat is found primarily in tributaries of the Mesilinka River. There are no known fish barriers to salmonid fishes on the main stem. Other species found either in the main stem or tributaries include burbot (*Lota lota*), sucker species (*Catostomus* spp.), sculpin species (*Cottus* spp.), and various Cyprinids including northern squawfish (*Ptychocheilus oregonensis*). Mature white and black spruce, aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), willow species (*Salix* spp.), and red alder (*Alnus rubra*) are the dominant trees and shrubs in the riparian zone. Logging has occurred to the stream bank of some of the tributaries and within sections of the main stem, and these activities have likely impacted fish habitat; however, the extent of these impacts in the Mesilinka watershed is undocumented.

The external control in this project, the Nation River, is located about 100 km south of the Mesilinka River, and also flows into the Williston Reservoir (Fig. 4). The river has a lake at its head and has a drainage area of 5880 km^2. Mean monthly flows in May, June, July, August, and September are 244, 293, 108, 40, and 27 $m^3 \cdot s^{-1}$, respectively. The Nation River tends to be more nitrogen (N) limited than the Mesilinka River because of the large lake, while both rivers are equally phosphorous (P) limited.

The potential for riverine fertilization to substantially enhance salmonid and grayling production is limited to those streams with: a) adequate juvenile recruitment; b) nutrient deficiencies; c) warmer streams with mean summer temperatures more than 10°C, preferably 12–15°C; d) abundant adult fish habitat; and e) high overwinter survival, the latter positively related to winter flow level and availability of pool habitat. The Mesilinka River is near ideal in physical habitat within its upper, middle, and lower reaches because it has sufficient flows, pools,

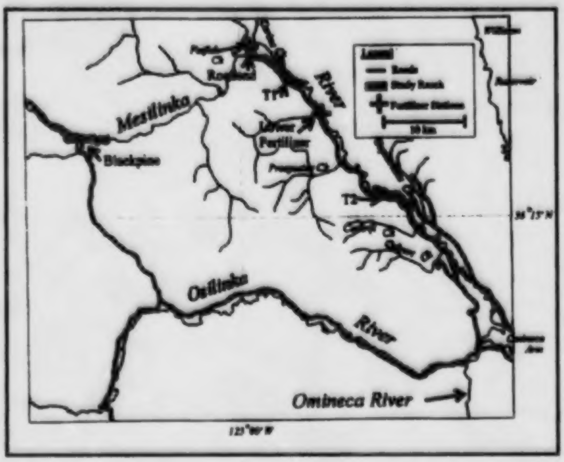

Figure 5. Location of control and fertilized reaches (N and P added), Mesilinka River.

and cover. Side-channel and tributary fish rearing areas are also sufficient. Mean monthly water temperatures in the Mesilinka middle and lower reaches during July and August (12–13°C) are within the lower acceptable range to augment salmonid growth.

Materials and Methods

The 1992–1995 (June–September only) sampling programs on the Mesilinka River (main stem and selected tributaries) consisted of sampling stream flows, water temperature, water chemistry, periphyton, benthic insects, and fish species. The Nation River was also sampled in a similar manner, but to a lesser extent. Three reaches were designated and individually sampled in the Mesilinka River: Blackpine (the experiment control reach), and T1 and T2 (fertilizer treatment reaches, in July and August 1994–95) (Fig. 5). These reaches are 7.5, 7.2, and 8.1 km in length, respectively, with wetted widths of about 35–40 m (August). They were chosen on the basis of accessibility and general similarities.

Mesilinka flows were obtained from the Water Survey of Canada. Tributary flows were derived from instream velocity measurements and stream area. Water temperatures were recorded with automated temperature data loggers (Tempmentor, by Ryan Instruments; and Hobo by Onset Computer Corp.) in each of the three reaches and in selected tributaries. Dissolved nutrient concentrations were sampled and measured as described in Johnston et al. (1990). Nutrient water chemistry variables measured were: low-level nitrate–nitrogen, ammonium–nitrogen, total dissolved nitrogen, low-level ortho-phosphorus (also called soluble reactive phosphorus, or SRP), and total dissolved phosphorus (TDP) on a biweekly to monthly basis. All of the above nutrients were measured in dissolved rather than particulate form. Nutrients, and in addition, total alkalinity, pH, and total dissolved solids (1992 only) were sampled on a monthly basis. Nonfilterable residue (NFR) and turbidity samples (indicators of water transparency) were collected weekly because of glacial turbidity in spring to early July. Handling and analysis of samples followed standard methods as in American Public Health Association (1985). Transparency was also measured *in situ* in the main stem using Secchi disk visibility.

Periphyton accrual, as measured by peak chlorophyll-a content, provides a useful indicator of the potential effects of nutrient stimulation (Perrin et al. 1987). Periphyton accrual was measured as described in Bothwell (1988) and Johnston et al. (1990). In this method, plexiglass plates, black in color, were bolted to concrete blocks using four stainless bolts per block. Styrofoam (30 cm x 30 cm x 0.6 cm) was fastened to the plexiglass plates with stainless steel wire. At each site the blocks were placed in a pool tailout at a depth of about 0.5 m below the water surface. Two 2-cm cores of periphyton (duplicate samples) were sampled at approximately 2- to 4-week intervals from the styrofoam substrata at each station, and each was analyzed as in Bothwell (1988). In each summer sampling season, 10 periphyton sampling blocks were installed from mid-July to mid-September in each of the three main stem reaches and in select tributaries. Styrofoam substrata were replaced after periphyton accrual peaked, or after 4–6 weeks (as in Johnston 1990). Thus, each set of chlorophyll-a determinations provided replicated accrual rates and peak levels in each of the three reaches.

Insect colonization baskets (cylindrical in shape, 22 cm in diameter by 13 cm in depth; 0.04 m^2 in area and 0.005 m^3 in volume, made of plastic) containing 3–6 cm diameter gravel were installed in riffle areas within the control and test reaches. For statistical purposes at least five baskets were installed at each reach. Baskets were surrounded by cobbles to minimize dislodgement, then left to colonize with insects for 6–8 weeks. At removal a Surber sampler (0.2-mm mesh net) was placed directly downstream from the basket, the gravel was scrubbed, and the contents of the basket (insects and detritus) were washed into the sampler. Based on earlier studies in other rivers, 80% of the peak abundance of insects is usually attained in 6 weeks and peak abundance attained at 8 weeks (Mason et al. 1973).

Fish Sizes and Abundance

During July and the first 2 weeks of August, fish were sampled in the three test reaches of the Mesilinka River, using standard angling techniques with barbless hooks. Fish lengths (fork-length) and weights were measured and recorded for catchable (mainly ≥ 20 cm) rainbow trout, Arctic grayling, bull trout, and mountain whitefish. Scales of rainbow trout and grayling were sampled for size-at-age determination. Scales were processed and aged as described in Ward and Slaney (1988).

Low conductivity confirmed that boat-shocking would be ineffective in the Mesilinka River, and therefore an alternative method for estimating fish populations was utilized, namely, underwater census (also referred to as snorkel surveys, or swim counts). Fish caught in the first 2 weeks of August (1992–95) were tagged with Floy tags (different colors per reach and per year) for subsequent population estimate studies using mark–recapture (visual) techniques. In the third week of August, highly standardized underwater counts were completed by a crew of six experienced divers using the method described in Slaney and Martin (1987). Water temperatures at the time of each swim were 11°C or higher because otherwise fish are found to be inactive and easily missed (Slaney and Martin 1987). Underwater census had been conducted previously by Fisheries Program staff in several trout streams in British Columbia including the St. Mary River (Slaney and Martin 1987), Adam River (Slaney et al. 1993; Toth et al. 1996a), and Big Silver Creek (Toth et al. 1996b). Systematic evaluations of underwater census techniques are found in Northcote and Wilkie (1963), Goldstein (1978), Griffith (1981), Gardiner (1984), Hankin and Reeves (1988), and Heggenes et al. (1990), which confirm the usefulness of the method, provided low temperatures (<11°C) were avoided for resident salmonids. In mark–recapture activities, fish are typically marked up to 2 weeks in advance of recapture to facilitate redistribution, as recommended by Vincent (1971).

Fish in the three Mesilinka reaches were counted by underwater census over a 5- to 6-day period. Within each of the three reaches, each float was replicated twice at random, within shore and mid-channel lanes. Tagged (i.e., recaptured fish) and untagged fish were recorded by species and by 10-cm-length category within each of the three reaches. Underwater census combined with mark–recapture techniques were then used to calculate correction factors for the underwater swim counts (as in Slaney and Martin 1987), because not all fish are detected in every swim. These correction factors provide a more accurate estimate of fish numbers, and can be used in subsequent years when mark–recapture estimates are not conducted. Complete details regarding methods (including calculation of correction factors, and methods used in the tributaries) are provided in Koning et al. (1995).

Nutrient Addition

Nutrients were added to the Mesilinka River (upstream from T1 and T2) in 1994 and 1995 (end of June to the beginning of September), by the addition of liquid ammonium polyphosphate fertilizer (10-34-0; % by weight $N:P_2O_5:K_2O$) and urea-ammonium nitrate (28-0-0; % by weight $N:P_2O_5:K_2O$). These fertilizers were added (via streambank-located tank and valve-controlled gravity-fed plastic hose) in sufficient quantity to provide an instream concentration of 5 $\mu g{\cdot}L$ dissolved inorganic P and 20 $\mu g{\cdot}L$ dissolved inorganic N. In 1994 both P and were added above T1, while only N was added above T2. In 1995, we added N and P fertilizers at both sites.

Results and Discussion

Complete data on 1992–95 area air temperatures, precipitation, main stem hydrology and water quality are found in Koning et al. (1995) and Paul et al. (1996, 1998). Pre-fertilization N (nitrate–nitrite N) and P (soluble reactive P) ranged from 5–26 $\mu g{\cdot}L$ N and from < 1–3 $\mu g{\cdot}L$ P. Target concentrations of N and P (in T1 and T2) were not always recorded or achieved in (July–August) 1994 and 1995. This was likely due to the poor injector system, based on gravity feed that frequently became plugged, and secondly due to rapid uptake by periphyton. A portion of the phosphorus will likely pool at depth in river sediments which will subsequently be available for release gradually throughout the year, but there is limited information on nutrient spiral-pathways in smaller (Newbold et al. 1981) and larger streams (Slaney, Rublee, Perrin, and Goldberg 1994). Total dissolved solids (TDS), pH, and total alkalinity were measured in 1992 but not in 1993. Mesilinka River values of TDS were 48–90 $mg{\cdot}L^{-1}$, pH 7.5–7.8, and total alkalinity 29–60 $mg{\cdot}L^{-1}$ (based on 1992 results). Over the summer, nonfilterable residue (NFR) (detection limit 4 $mg{\cdot}L^{-1}$) at main stem sites ranged from 4–30 $mg{\cdot}L^{-1}$, and turbidity from 0.1–5 NTU. Secchi disk depths (visibility from the water surface) ranged from 0.7–2 m. These values (1992–93) are typical and suggest sufficient water clarity exists for periphyton growth in areas with available nutrients.

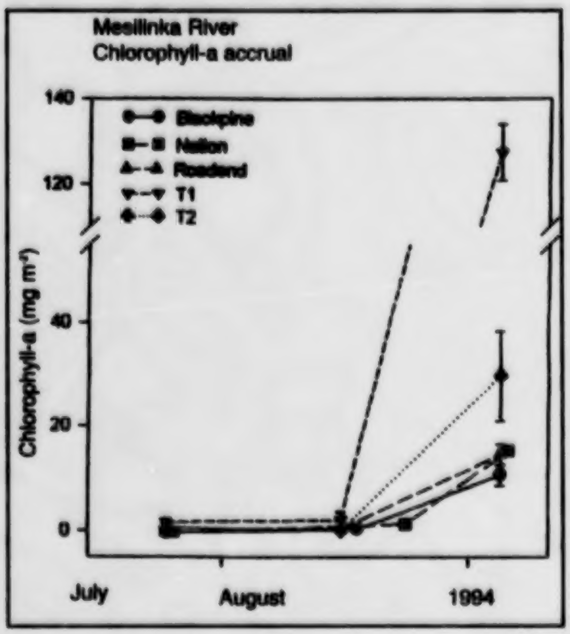

Figure 6. Chlorophyll-a accrual, Mesilinka River 1994 (year one of nutrient addition). Blackpine and Roadend are two upstream control reaches in the Mesilinka River; the Nation River serves as an external control. T1 and T2 are the two treated reaches.

Periphyton accrual, based on measurement of chlorophyll-a content (mid-July to mid-September), peaked at approximately the same rate and low magnitude (7–16 mg·m^{-2} over 8 weeks) at Blackpine, T1, and T2 in pre-fertilization years 1992–93. This suggests that the three sites are similar for comparison of impacts from input of nutrients in future years. Nation River results were similar. Addition of N and P in 1994 resulted in significant increases in periphyton accrual in the treated reaches T1 and T2 (Fig. 6). Results in 1995 were similar, except in T2, where accrual remained low, possibly due to increased periphyton grazing by benthic insects.

Assessment of the colonization of artificial substrate by aquatic insects provides a useful indicator of the response of salmonid food chains to enrichment, and is thus a good indicator of the potential for trout growth as documented elsewhere (Slaney and Ward 1993). In 1993 (pre-fertilization), mean benthic invertebrate biomass at Blackpine, T1, and T2 varied from 0.5 to 17 g·m^2. Invertebrate biomass in T2 increased from 0.5 to 6 g·m^2 between 1993 and 1995. Biomass in T1 remained between 4 and 7 g·m^2; and biomass in the control reach (Blackpine) decreased from 1993 to 1995 (by about 60%). By comparison, samples from the Salmon River on Vancouver Island weighed between 5 and 12 g·m^2 at un-enriched sites, and up to 38 g·m^2 at enriched sites (Slaney, Ashley, Wightman, Ptolemy, and Zaldokas 1994). Numbers of individual invertebrates in the Mesilinka samples followed the same trends as in the biomass. Based on preliminary observations of the Mesilinka samples, most common were various genera from the orders Plecoptera (Chloroperlidae family), Ephemeroptera (Baetidae family), and Diptera (Tipulidae and Chironomidae families).

Fish Sizes and Abundance

Fish species for which we recorded size and age data include rainbow trout, bull trout, Arctic grayling, and mountain whitefish. In total, in each of the 4 years about 500–700 fish were caught by methods that included angling, electrofishing, trapping, and gill netting. Rainbow trout were the most numerous fish species sampled, based on the type of sampling methods used (mainly by angling). In actual numbers, mountain whitefish (a species much less catchable by angling gear) were by far the most abundant fish species, based on underwater census.

Length (fork-length) and weight records for Mesilinka and Nation river salmonids indicate bull trout range in size up to 800 mm and 4.6 kg (average length less than 400 mm). Mountain whitefish were the smallest salmonid species with fork-lengths mostly in the 225 to 300 mm range. Most rainbow trout were in the 200–350 mm range, and most Arctic grayling ranged from 250 to 390 mm. Age composition of rainbow trout and Arctic grayling were similar between the two rivers. In both rivers age 3+ rainbow trout and age 4+ Arctic grayling predominated. Overall, there was a tendency towards slightly greater sizes and growth rates per year for rainbow trout and Arctic grayling in the Nation River, and this is probably a result of warmer summer water temperatures there. Length and weight data in 1994–95 (Arctic grayling and rainbow trout, age 3+ and 4+) did not show significant increases over the pre-fertilization data (from 1992–93), with the exception of 1995 (year two of fertilization), when for the first time, size at age of rainbow trout (4+ fish only) increased significantly in the treated reach, T2, compared to the control [34% increase in weight, analysis of variance (ANOVA), $p<0.05$]. The response of the fish populations to fertilization may initially lag in the Mesilinka, because the cooler temperature regime of the Mesilinka (<12°C mean monthly summer temperatures) can be expected to moderate both the response and potential benefits to a fishery.

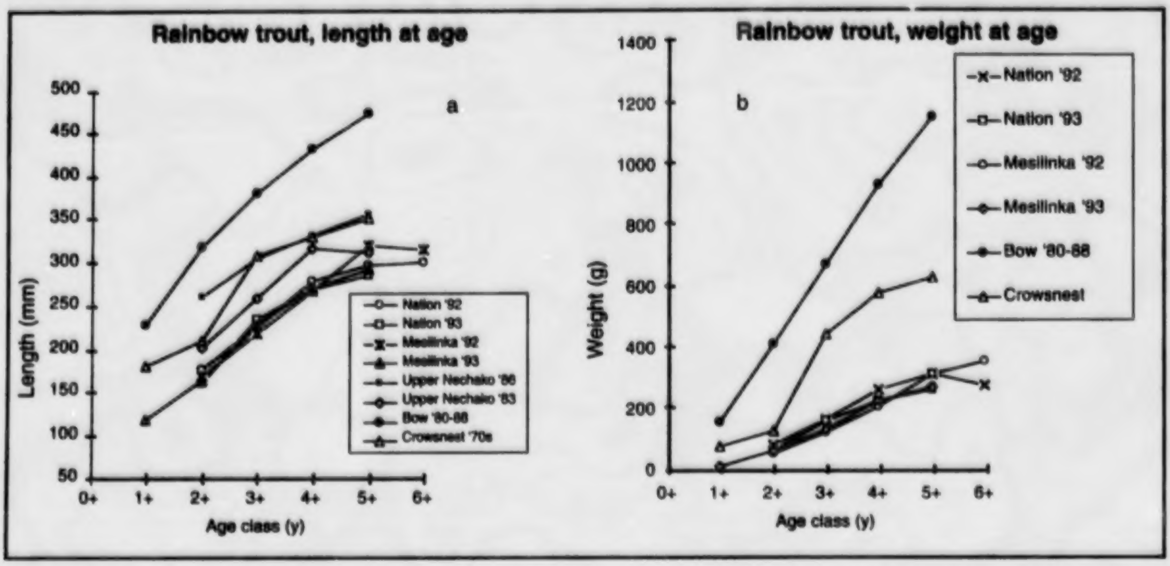

Figure 7. A comparison of rainbow trout, (a) length at age, and (b) weight at age in various rivers. The Mesilinka values (1992, 1993) are prior to fertilization. The Bow (below Calgary) and Crowsnest (in the 1970s) rivers are located in Alberta and are subject to treated municipal wastewater discharge. The Nechako River is in north-central B.C. and is quite similar to the Mesilinka River; the difference between the two years (1983, 1986) reflects a sport fishing closure in the river after 1983.

Comparisons to Other River Systems

Length- and weight-at-age (Fig. 7a, 7b) of rainbow trout size from the Mesilinka and Nation rivers were compared to those in the upper Nechako River (1983 and 1986, *in* Slaney 1986), the Bow River below Calgary (Courtney and Fernet 1991), and the Crowsnest River, flowing east out of the Rocky Mountains in southern Alberta (Alberta Fish and Wildlife Branch files, Lethbridge, Alberta). The Nechako is a northern river quite similar in geographic character to the Mesilinka and Nation rivers but with an oligo-mesotrophic nutrient regime (Slaney 1986). Water temperatures in the upper Nechako River (data on file) during the summer are higher than those in the Mesilinka, and similar to or slightly higher than those in the Nation River (Slaney 1986). The Bow River below Calgary contains ideal rainbow trout habitat combined with a very high nutrient regime due to treated wastewater effluent being discharged by the city. Similarly, the Crowsnest River (in the 1970s) also received treated municipal wastewater but to a lesser extent. Mean monthly water temperature in the Bow River in August ranges from 15–16°C (1975 data, *in* Culp et al. 1992), which is similar to the Nation River. Climate and geography in the Bow and Crowsnest systems are similar to those of the Mesilinka and Nation rivers. Clearly, rainbow trout in the Mesilinka and Nation rivers are substantially smaller than those in the upper Nechako, Bow, and Crowsnest rivers. Thus, data from the Nechako, Bow, and Crowsnest are supportive of the potential of low-level nutrient addition to enhance fish growth in cold-water, oligotrophic, interior systems such as the Mesilinka and Nation rivers.

Fish Abundance Estimated by Underwater Census and Mark–Recapture

Based on underwater counts (Fig. 8), the density (number of fish per hectare) of all four fish species in T1 increased in year two (1995) of fertilization. (Note that correction factors based on mark–recapture results have not been applied in Fig. 8.) Recognizing that substantial variability in fish density occurs from year to year, the results nevertheless are supportive of beneficial responses in the fish populations. In particular, these initial results for rainbow trout and mountain whitefish suggest increases over pre-fertilization years of at least two-fold and five-fold, respectively. Due to precipitation-induced turbidity problems, underwater census results are not available for T2 in 1995, and we completed only a single unreplicated swim result in the control reach (the replicated T1 and single control swims were carried

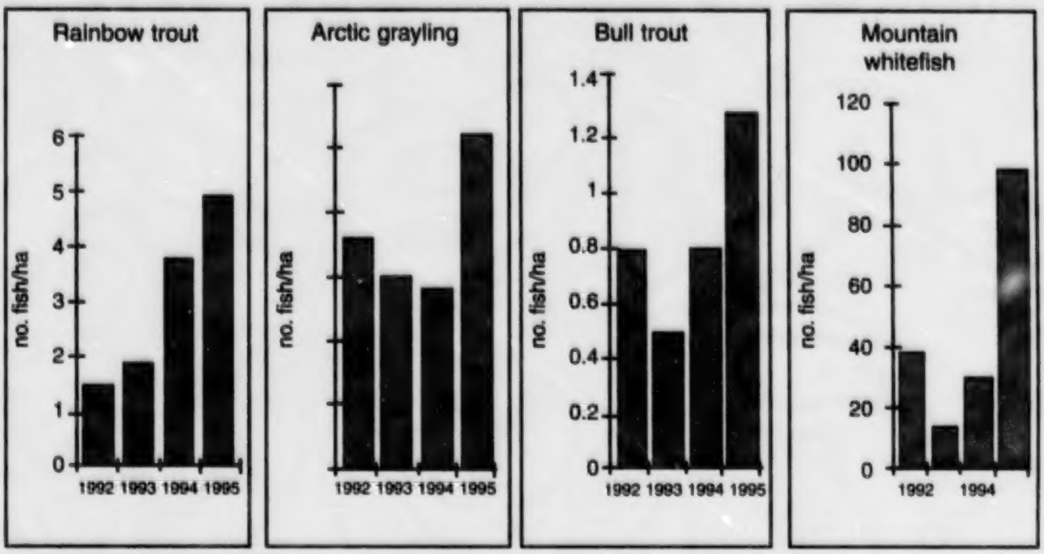

Figure 8. Mesilinka River swim counts in treated reach, T1. Numbers are based on census by six swimmers with mask and snorkel floating in line (in six lanes) in a downstream direction and expanded to account for fish not counted between lanes. No correction factor has been added to account for fish not observed by the swimmers, e.g., fish hidden within woody debris, and/or missed due to high velocity sections.

out prior to elevated turbidity levels). The unreplicated control reach swim result in 1995 combined with the previous 3 years of data do not show the same rainbow trout and mountain whitefish results as evident in T1 in 1995. Further comparisons, confirmations, and conclusions await several additional years of data collection.

The observed increase in rainbow trout and mountain whitefish numbers in T1 (and not in the control reach) after only 2 years of fertilization is likely attributable to fish attracted into the river from downstream areas, such as the reservoir, rather than to within-reach increases (recruitment). Recent observations in Sweden suggest that fish from downstream areas are likely being attracted by food odors, possibly responding to chemicals released by the periphyton, or periphyton-based community. Attracting fish from other reaches allows for greater overall productivity in the river.

Migratory adfluvial (lacustrine) stocks in the embayment area may benefit from the proposed river fertilization project because of stimulation of the riverine food chain and resultant spiralling of nutrients downstream to the embayment. Results of this kind were found with small-scale nutrient treatment at a stream and embayment at a northern Swedish reservoir, where increased zooplankton and benthic fauna benefited Arctic char and European grayling, respectively (Milbrink and Holmgren 1981). In addition, attraction of rainbow and bull trout into the Mesilinka River from the embayment could be substantial, based on the Milbrink and Holmgren (1981) study. The substantial size of some of the larger bull trout (3–5 kg) caught in the Mesilinka River (1992–95) suggests that this may already be occurring (i.e., the bull trout spend winter to spring months in the reservoir, similar to adult bull trout and rainbow trout in the Skagit River–Ross Reservoir, a system in southern B.C.).

Preliminary Conclusions

Results of field work in 1992–93 confirmed the suitability of the Mesilinka River for whole-river fertilization to mitigate some of the earlier impacts of reservoir flooding of the lower riverine reaches. The Mesilinka River contains sufficient flow and

abundant adult-rearing habitat. Water temperatures are low but within the acceptable range for salmonid growth (especially for char and grayling growth), and water quality is adequate. Turbidity and suspended sediment levels can persist into mid-July, but transparency (more than 1 m) appears to be sufficient to initiate river fertilization in late June to early July. Suitable juvenile fish habitat is available in the tributaries and in the side-channels of the main stem. Underwater counts indicate the Mesilinka River has demonstrated there are sufficient numbers of rainbow trout, Arctic grayling, bull trout, and mountain whitefish (up to 31, 31, 9, and 310 fish/km, respectively) to continue and support fertilization as a mitigation option. Based on capture methods used, most catchable-sized rainbow trout and Arctic grayling are small and less than 30 cm. Results of nutrient addition (1994–95) show impacts on the food chain (increased periphyton growth and benthic invertebrate biomass and density) within the treated reaches, with an apparent trend towards increased fish biomass and numbers. These results await further confirmation, with expansion of fish populations as a likely outcome (Milbrink and Holmgren 1981).

Application of the Mesilinka results and of those obtained previously from the Keogh and Salmon rivers in B.C. are directly applicable to restoration of stream impacts due to past timber harvesting activities, in particular where harvesting to the stream bank has reduced the quantities of organic debris and food materials (allochthonous materials) entering the stream, or where degraded habitat results in higher than average metabolic energy required to maintain basic fish survival. Recent studies using stable isotope analyses have identified the important role of salmon carcasses in returning nutrients and food materials to anadromous streams (Schuldt and Hershey 1995; Bilby et al. 1996) and thus the negative consequences on stream trophic levels where due to overharvest or degraded habitat, numbers of returning salmon are far below natural levels. Adding nutrients in such situations aids in returning streams to pre-impact fish productivity levels.

Stream fertilization therefore is one more tool in the restoration toolbox. It does not take the place of hillslope stabilization, riparian rehabilitation, fish access improvements, input of large woody debris and boulder placements in streams, as well as other restoration techniques that are available, but in many streams it is an appropriate technique, to be used with others, to accelerate the restoration process.

New Developments

Recent developments (1995) in stream fertilization projects conducted by the B.C. Ministry of Environment are: a) the use of a flow-proportional liquid fertilizer injector system; and b) in smaller streams the use of slow-release solid fertilizer briquettes. The currently used manually operated liquid metering systems require daily adjustment to maintain constant application rates; hence, their cost effectiveness is reduced and the high maintenance requirements limit the application of this technique to easily accessible sites. A lab-scale automated system was developed to meter liquid fertilizer into rivers and streams (Ward and Associates 1995a), and was subsequently field tested (Ward and Associates 1995b). It was designed to operate in remote locations with minimal maintenance, and to automatically adjust metering rates to variations in river stage to maintain a constant low-level nutrient addition. This automated system significantly reduces the labor requirements of stream fertilization and provides a more accurate injection of liquid fertilizer, which should reduce the variability of our results.

Slow-release solid briquettes containing fertilizer (N and P) have been developed by Vigoro Chemicals (Winterhaven, Florida) at our request as an alternative to the use of liquid fertilizer. Although initially more expensive to purchase, their advantage lies in ease of use. The briquettes, which are the size of small spheres (diameter 3 cm) or cylinders (length 4–5 cm), are measured out (by weight) and added to a stream (in low stream velocity locations) at the beginning of the summer growing season, during the summer low stream flows. They slowly release nutrients (N and P) on a continuous basis over several months. During the current testing period, water quality, periphyton accrual, invertebrate biomass, and fish numbers are being monitored (Mouldey and Ashley 1996). The intention is to use slow-release briquettes in smaller systems where regulating very small amounts of liquid fertilizer injection is too labor intensive and inaccurate.

The flow proportional injectors were used for the first time in the summer of 1995 in Big Silver Creek (summer flows 8–42 m^3/s), which flows into Harrison Lake, and the Adam River (summer flows 8–17 m^3/s) located on Vancouver Island. Results suggest a highly uniform and highly productive growth of periphyton downstream of the fertilizer additions (Toth et al. 1996a, 1996b). The slow-release briquettes were also field tested for the first time in the summer

of 1995, at about 10 locations, including tributaries of Big Silver Creek and the Adam River. Water quality analyses and positive periphyton accrual at downstream monitoring stations confirm the ability of the slow-release briquettes to produce the expected results over the growing season (Mouldey and Ashley 1996).

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Protecting British Columbia's Fish Streams: A Sierra Legal Defence Fund Perspective on Fisheries and Forestry Lawsuits

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Abstract

There is a large and growing body of scientific evidence that supports the view that the logging of timber and associated activities, particularly road building, can adversely affect the productivity of fish habitat in areas where these activities are undertaken if they are not carefully planned and carried out. This scientific evidence is supported by recent, anecdotal studies conducted in British Columbia (B.C.) between 1992 and 1997, which examine the impacts of contemporary forest harvesting activities on British Columbia's fish habitat. These studies clearly show that, despite the B.C. and federal governments' best efforts to regulate the forest industry, through the introduction of tough environmental laws, and effect change in the way forestry is conducted around fish streams in B.C., fish habitat is still not getting the protection it deserves. This has caught the attention of the general public and initiated a public outcry for stronger environmental standards and rigorous enforcement of existing laws. This paper examines the regulatory regime as it applies to timber harvesting and fish habitat in B.C., and the reasons why these laws do not go far enough to protect this valuable resource. A non-government, citizen-led approach to forcing government and industry to change the way they do business in B.C. is proffered and evaluated for its effectiveness. Many of the opinions expressed in this paper are those of Sierra Legal Defence Fund, and these views may not be shared by other stakeholder groups.

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Introduction

There is a large and growing body of scientific evidence that supports the view that the logging of timber and associated activities, particularly road building, can adversely affect the productivity of fish habitat in areas where these activities are undertaken if they are not carefully planned and carried out (Ringler and Hall 1975; Chamberlin 1982, 1988; Salo and Cundy 1987; Carr 1985; Murphy et al. 1986; Cederholm and Reid 1988; Hartman and Scrivner 1990; *see* Marcus et al. 1990 for a comprehensive review and indexed bibliography).

The Pacific Northwest is home to several large, commercially important salmon rivers such as the Stikine, Skeena, Nass, and Fraser in British Columbia (B.C.) and the Columbia in Washington State. These river systems produce millions of fish annually, which are harvested both commercially and recreationally from Alaska to Oregon. In addition, hundreds of smaller rivers and thousands of tributary streams that drain the seaward side of the Coast Range mountains also produce their own separate, genetically distinct races of fish (Beecham et al. 1995; Miller et al. 1996, Beecham et al. 1996) that add to the total production of salmon on this coast. In 1981, it was estimated that commercial salmon and sport fisheries produced an average catch of about 25 million fish from about 2500 known salmon streams in B.C. alone (Toews and Brownlee 1981). In addition to the five species of Pacific salmon, B.C.'s rivers and streams are home to four species of trout and five species of char (McPhail and Carveth 1993), which are of importance to recreational fishers.

Each species of salmon and trout have distinctly different habitat requirements for spawning and rearing. For example, chinook salmon (*Oncorhynchus tshawytscha*) typically spawn and rear in the large mainstem rivers, while the smaller mountain streams provide critical spawning and rearing habitat for coho salmon (*O. kisutch*), steelhead (*O. mykiss*) and cutthroat trout (*Salmo* spp.). These fish once were the mainstay of the Canadian and northwestern United States commercial and recreational fishing industries, but stocks have become severely depleted in B. C., California, Oregon, and Washington. Many wild salmon and steelhead runs have been petitioned for listing under the United States *Endangered Species Act*.

Declines in the wild stocks of salmon and trout on the west coast of North America can be largely attributed to habitat losses and damage caused by land use activities such as logging, agriculture, industrial, and urban development, as well as hydroelectric power generation. However, in the largely unpopulated, non-industrialized, and mountainous coastal regions of B.C., logging is the primary land-use activity leading to fish habitat loss. In British Columbia, the forest sector is the second largest sector in the B.C. economy after the finance, insurance, and real estate sector (Mascall 1994).

There is legislation in Canada specifically designed to protect fish and fish habitat from the impacts of industrial and urban development. However, in B.C. an active policy of prosecuting the major contributors to the problem of fish habitat loss is severely lacking. Over much time there has been built into the present political system a special relationship between the regulators (the resource agencies) and the regulated (logging companies), and this has resulted in the devastation of our forests and fish habitat with responsibility being attached to no one. There is a relaxed regulatory atmosphere. Inspections are often conducted by appointment. Government regulatory staff see themselves as trouble-shooters and problem-solvers on behalf of the logging industry. There appears to be a naive willingness to accept the client's version of events and circumstances. Senior government and political officials are too frequently susceptible to the pleadings of corporate executives. Quite simply, in British Columbia there is not the political will to charge powerful corporations with criminal environmental offenses. In addition, the special relationship that has evolved between the resource agencies and the logging companies has complicated matters. Provincial Crown prosecutors often refuse to proceed with charges when they are laid if there are grounds to believe that the offender may successfully argue a defense of either officially induced error (the government permitted the offence to occur) or one of due diligence (that by following resource agency prescriptions the appropriate steps necessary to prevent the offence from being committed were followed).

Timber Harvesting And Fish Habitat Loss

The clear-cut: the most environmentally ruinous form of tree harvesting is the accepted method of logging in British Columbia. Even the B.C. *Forestry Act* is specifically written to sanction this kind of tree harvesting. Clear-cutting entails the removal of virtually every single tree from a designated plot of land. Nothing is left standing, except, perhaps, an eagle nesting tree.

Historically, the standard size of individual clear-cut plots in B.C. have been as large as 120 ha (296 acres). By itself, a clear-cut of this size would not be an ecological disaster. However, for years the B.C. Ministry of Forests (MOF), the stewards of our forested land, have allowed forest companies to cut individual blocks of clear-cuts adjacent to one another. Thus, in B.C. today we have what are termed progressive clear-cuts that can extend for miles. This has resulted in the complete removal of all forest cover from river valleys and entire mountains. Clear-cutting on such a grand scale is environmentally devastating.

One of the major environmental impacts associated with clear-cut logging is the destruction of fish habitat. In virtually every area being harvested there are rivers and streams that support salmon and/or trout and other fish species. These streams are being systematically destroyed by current logging practices (c.f. Tripp et al. 1992, 1993; Tripp and Grant 1993; Tripp 1994).

Long-term studies of fish–forestry interactions in the Carnation Creek watershed on Vancouver Island (Chamberlin 1988) indicate that forestry activities there resulted in reduced survival of coho and chum salmon populations in that stream by 50% and 25%, respectively. This occurred despite the fact that much of the logging in the Carnation Creek was controlled to minimize stream impacts.

About one-quarter of the watershed was logged in a manner consistent with the way things had been done in British Columbia for decades. The rest of the watershed had controlled logging, which included new techniques such as leaving a protective buffer of trees (streamside management zone) and clearing logging debris from channels concurrent with harvesting activities. The study found that the greatest impacts on the salmon resources in Carnation Creek arose in the uncontrolled logging portion of the watershed where increased streambank erosion and sedimentation rates, downstream transport of woody debris left on streambanks, loss of large stable large organic debris (LOD; i.e., logs, root masses, and other naturally occurring woody debris) and altered stream flow patterns were the most severe effects.

Timber harvest and associated road building lead to increased rates of erosion into stream channels resulting in altered substrate composition within channels (Sullivan et al. 1987). Channel disruption, decreased rooting strength, loss of stable debris from channels, and elevated sediment loading decrease the stability of channel morphology and the stream substrates (Sullivan et al. 1987; Chamberlin 1988). These changes in the environment and structure of stream systems have profound implications for stream ecosystems.

Timber Harvest

Removal of the forest canopy in clear-cut areas alters the natural cycle of water from soils to trees to the atmosphere and back to the soil such that the entire hydrological regime of the land can be adversely impacted (c.f. Clayoquot Sound Scientific Panel 1995). This is because removal of the forest canopy enables most of the rainfall to reach the ground, rather than have a portion of the rainfall intercepted by the canopy for subsequent evaporation, whereas intact rainforests naturally absorb rainfall and release water slowly through natural processes. Once tree cover is removed, the land can no longer hold back the moisture and large amounts of water are released very quickly causing increased surface runoff and higher peak flows in streams (Chamberlin 1988). These higher streamflows cause otherwise stable, armored streambed particles (gravel, cobble, and boulders that do not normally shift under the normal range of flow conditions found in a given stream) and LOD to mobilize causing the stream to sluice out. In addition. the exposed soils in the surrounding drainage can begin to erode and/or slump, transporting more sediments downslope and depositing them into watercourses. Steep, unstable slopes are especially prone to erosion and landslides.

Cross-stream falling and yarding during clear-cutting usually results in logging debris being deposited within the annual high water mark of streams and ephemeral channels draining virtually every cut block. During high flows, this material can be transported downstream, ripping apart streambanks and beds, and forming debris dams, which can cause streams to breach their natural channels or cause mass wasting and torrenting events when the dams themselves breach. In addition, the yarding of trees across streams can result in localized areas of streambank destabilization in situations where the boles and butt ends of trees impact directly on the streambanks as they are being yarded across.

Clear-cutting of streambanks and the attendant decay of stabilizing root systems has a profound impact on streambank stability. Within a year following logging, streambanks begin to deteriorate and collapse into the main channel adding tonnes of fine and coarse sediment and woody debris to the system (personal observation). Root wads of once living trees lining the stream become undermined and fall

into the main channel as well. These are then transported downstream during high flows and contribute to further streambank destabilization either by ripping apart the banks as they smash into them while tumbling downstream or by causing or adding to large debris jams.

Finally, trees along the boundary edges of cutblocks and those left in inadequately planned streamside management zones become exposed to high winds that are normally buffered by an intact forest canopy and they quite often blow down. Many streamside protection zones have blown down completely causing massive bank destabilization and adding excess woody debris to the channel.

Road Building

Clear-cutting requires an extensive network of roads to access and remove timber. This results in miles and miles of productive forested land being laid open like a gaping wound on the land, exposing otherwise stable soils to erosion.

Road building results in a major alteration of the natural patterns of water runoff from forest lands as permanent and ephemeral streams draining mountainsides are bridged, culverted, and/or diverted. It has been a common practice in the past to collect water from several drainage channels draining a slope and divert it, via ditches, into a single recipient stream channel. This causes the natural stream flows in the receiving channel to increase markedly, resulting in massive uncontrolled erosion. In addition, exposed mineral soils from within ditches and slumping soil from unprotected, unstable road cutslopes is washed out of the ditches, carried downstream, and deposited into the receiving stream.

Road surfaces act as conduits for surface runoff water running downhill during periods of heavy rainfall, especially in areas with inadequate ditching or cross drainage. Water running along roads mobilizes the fine sandy material on the road surface and transports it downhill to the nearest watercourse blanketing the streambed and filling in pools. Road surface erosion is a major problem in previously logged areas where abandoned roads are not being maintained and ditches have filled in from cut-slope soil creep.

Improperly planned and engineered roads and poor road construction and/or maintenance have caused and continue to cause numerous landslides (Northwest Hydraulic Consultants Ltd. 1994). This further reduces the amount of productive forest land and deposits tonnes of sediment and debris into rivers and streams.

Fisheries Management and Logging in B.C.

A Historical Perspective

Until recently, all logging in British Columbia was conducted pursuant to the provincial *Forest Act*, which is administered by the provincial Ministry of Forests. Unfortunately, the *Forest Act*, with the exception of some minor administrative penalties, contains neither offense nor penalty provisions for protecting fish habitat from destructive logging practices.

Through agreement with the MOF, the Federal Department of Fisheries and Oceans (DFO), the B.C. Ministry of Environment, Lands and Parks (MOELP), and the forest industry, an inter-agency referral system was initiated in 1956 (Toews and Brownlee 1981). This process was implemented to provide for increased protection to aquatic habitat from the ravages of logging by initiating the inclusion of restrictive clauses in logging contracts for logging in and around streams. the DFO and MOELP were provided with the opportunity to review 5-year development plans, management and working plans, and pre-harvest silviculture prescriptions submitted by the companies to MOF to assess likely impacts of the proposed logging plans on fish habitat. Where impacts to fish habitat were likely to occur, DFO and MOELP could proffer changes to the proposed logging plan to minimize the potential impacts. However, the effectiveness of this procedure has, to this day, been limited by inconsistent application and by a lack of enforcement due largely to the sheer volume of cutting activity, coupled with the limited response capability of the various resource agencies.

The DFO and MOELP do not have the human or financial resources to do an adequate job in this regard (Louis Tousignault, Director, DFO, Pacific Coast Region; Toby Vigod, Legislation Department, B.C. MOELP, personal communication). For the most part, ministry line staff rely on information about the fisheries resources in areas of proposed timber harvest, which is supplied to them by the logging companies. However, we have found that much of this information is lacking in sufficient detail for the resource agencies to make sound, informed decisions regarding how the proposed cutblocks should be harvested to protect fisheries resources. In addition, because of budgetary and staffing constraints, resource agency officials are unable to verify (i.e., ground-truth) the information they receive.

We have also found that in some cases, there is a general lack of cooperation between the resource agencies when something does go wrong. For example, in one case involving a December 1992 landslide caused by logging in the Miller Creek watershed on the Queen Charlotte Islands, both DFO and MOELP agreed that a prosecution of the logging company responsible could result. Problems arose, however, when the DFO placed responsibility for the necessary investigation with the provincial enforcement agency. The provincial authorities, on the other hand, maintained that the DFO should be the lead agency in the investigation. As a result of this conflict, neither agency acted for over a year. The case was finally brought forth a year later in January 1994, but only as a result of a private prosecution initiated by the Sierra Legal Defence Fund on behalf of the Steelhead Society of B.C.

Establishment of the B.C. Fisheries/Forestry Guidelines

The results of the Carnation Creek Study (Chamberlin 1988) and other studies conducted throughout the Pacific Northwest (Cederholm and Reid 1987; Hall et al. 1987) and Alaska (Gibbons et al. 1987), show a clear connection between clear-cut logging and its associated road building and damage to fish habitat. In B.C., this fact was also recognized and accepted by the resource agencies and by the logging companies. It was also recognized that something had to change or we would risk losing what little unaltered fish habitat remained.

To facilitate the referral process and to ensure that fish habitat was afforded a high degree of protection from logging activities, the MOF, DFO, MOELP, and the Council of Forest Industries (COFI, the agency representing all of the major logging companies operating in the province) formulated a set of guidelines to integrate fisheries and forestry resource management in coastal British Columbia (B.C. Ministry of Forests et al. 1988, 1992, 1993). The aim of the guidelines was to "improve the performance and effectiveness of the consultative process required for fish habitat protection and concurrent forest harvesting management".

The Coastal Fisheries/Forestry Guidelines (CFFG) (B.C. Ministry of Forests et al. 1988, 1992, 1993) addressed habitat requirements for fish and what steps should be taken by the forest industry to protect fisheries values. A stream-reach classification system was developed, which identified a range of fisheries habitat values ranging from Class I (Class A in later editions = highest value) to Class IV (Class C in later editions = lowest value). Accompanying this was a series of operating guidelines for all of the major forestry operations, which are to be implemented in accordance with stream-reach class objectives.

The operating guidelines cover topics such as road construction and maintenance, falling and yarding, and the post operational phase of logging including silviculture. Essentially, for every one of these activities, there was a series of guidelines to be followed to protect fish habitat.

One important aspect of the CFFG was the introduction of Streamside Management Zones (SMZ) on all Class I and II streams. Streamside Management Zones were designed to protect streams by preventing logging companies from operating too close to fish-bearing streams. The guidelines provided for buffer strips, which are stands of trees left along the streams to protect the banks and to provide a future source of LOD. Further, within the SMZ, only selective logging that did not involve the use of heavy equipment was permitted.

To ensure the guidelines were being complied with by logging companies, a monitoring program was to be set up to assess the degree to which fish habitat objectives were being achieved, and to determine the efficiency of the guidelines in reducing and resolving conflicts and making management decisions in the field.

The CFFG were first implemented in 1988 (B.C. Ministry of Forests et al. 1988). At that time, the Ministry of Forests was to take the lead role in the implementation and monitoring of the effectiveness of the program. However, due to the limited response capability of its staff, the MOF, through a Letter of Understanding transferred these responsibilities to industry (Ken Matthews, Operations Manager, B.C. MOF, Port Alberni Forest District, personal communication). The logging companies were, in effect, to police themselves and as a result government enforcement agencies were, for the most part, left out of the picture.

The CFFG were to be the ultimate solution to ensure the protection of fish habitat from damage caused by poor logging practices. It was thought that this would result in not having to use enforcement of environmental legislation to achieve the same result.

The Tripp Report

In early 1991, 3 years after the introduction of the CFFG, MOELP retained a private consulting firm, D. Tripp Biological Consultants Ltd., of Nanaimo B.C., to evaluate logging operations on Vancouver Island to determine the degree of compliance with the guidelines. Twenty-one randomly selected cut

blocks were audited, all of which had been logged after 1988.

Tripp et al. (1992) found that, for the most part, logging companies were not complying with the guidelines. He found that there was at least one major or moderate impact to fish habitat in at least one stream on every single cut block examined, an alarming statistic given that over the previous 5 years more than 6000 cutblocks had been logged along the B.C. coast. Half of the impacts were to Class I and II streams. In all, 34 (64.2%) of the 53 streams examined had been damaged by negligent logging practices. Damage included streams filled in with gravel, collapsed and eroded streambanks, and streams filled with logging debris. It was further determined that all of the observed impacts could have been avoided if the guidelines had been followed.

This report, known as the Tripp Report, indicated that habitat destruction was caused by poor gully management, inadequate drainage control on logging roads, failure to provide a leave strip of trees along streambanks, misclassification of streams, and in over 25% of the streams surveyed, failure to even classify streams at all. Significantly, the report makes it quite clear that the government regulatory agencies were in many ways responsible for these negligent logging practices in that they often failed to invoke pre-harvest prescriptions that were specific and enforceable and failed to conduct adequate on-site inspections both before and during logging activities.

Similar results, with varying degrees of compliance with the CFFG, were reported following subsequent, like studies conducted in 5 other forest districts around the province (Tripp et al. 1993; Tripp and Grant 1993; Tripp 1994). For example, in the North Coast, Kalum, and Sunshine Coast forest districts Tripp (Tripp et al. 1993; Tripp and Grant 1993) found that one-half to two-thirds of the stream reaches with fisheries concerns showed a major or moderate impact of some type caused by failure to follow the guidelines.

Activities that result in "the harmful alteration, disruption, or destruction of fish habitat" are prohibited under the federal *Fisheries Act*. However, no charges were ever laid in any of the situations brought to light in the Tripp Reports. Some preliminary on-site inspections were conducted on the sites with the worst damage and in some cases requests were made that remedial measures be undertaken to repair some of the damage. Otherwise, no other punitive actions have been taken.

Following the release of the Tripp Report, the MOF ordered all logging companies operating on Vancouver Island to re-visit all cutblocks logged since 1988 and prepare audit reports identifying areas of non-compliance with the CFFG. In addition, the MOF, MOELP and DFO were to conduct their own independent audits of 10% of these cutblocks. The findings of a selection of these audits were detailed in an internal government document (Martin 1993).

This document, dated August 6 1993, pointed out that virtually all major aspects of the CFFG still had a low level of compliance. Problem areas included poor road construction and maintenance, which resulted in damage to fish habitat and failure to leave sufficient mature timber in streamside management zones to protect streambanks for Class I and II streams. Importantly, this document also made it obvious that both federal and provincial resource agencies had approved these inadequate logging prescriptions that lead to damage to fish habitat.

On June 15 1995, the B.C. government enacted the new *Forest Practices Code of British Columbia Act* (the Code), which is being promoted (by the B.C. government) internationally as a model for forest management that is unparalleled anywhere in the world. The Code provides for penalties (fines up to $1 000 000 or 3 years imprisonment or both for a first offence) for carrying out forest practices that result in damage to the environment [section (s) 45(1) of the Code]. The Code incorporates and builds on many of the aspects of the CFFG with the intention of making the guidelines legally binding. However, a close examination of the Code reveals that there are few, if any, provisions that offer any greater protection for fish habitat in areas where timber harvesting is being proposed or undertaken (see below).

The Regulatory Regime

The Forest Practices Code of British Columbia Act *(FPCBCA or Code)*

Contrary to popular belief and the declarations of the B.C. government, the new Forest Practices Code does not provide adequate protection for fish habitat on forest lands in B.C. While the Code does provide for penalties (fines up to $1 000 000 or 3 years imprisonment or both for a first offence) to persons carrying out forest practices that result in damage to the environment (s. 45(1) of the FPCBCA), it contains the proviso that there is no contravention

of this provision of the Code if the person or persons conducting these activities is(are) acting in accordance with an operational plan or permit issued under the FPCBCA or its regulations. In addition, the term damage to the environment is not defined.

The Code also provides for additional fines ranging from $5 000 to $500 000 and/or imprisonment ranging from 6 months to 1 year (s. 143 of the Act). In addition, there are substantial administrative penalties (FPCBCA, Administrative Remedies Regulation, B.C. Reg. 166/95) to persons failing to comply with their operational plans and the terms and conditions of those plans. Operational plans include long-term (5 or more years) forest development plans, logging plans, and silviculture prescriptions (which are cutblock specific), access management plans (i.e., roads and road maintenance), and range use plans.

Approval of operational plans and permits as they relate to timber harvesting on Crown land in B.C. rests with the MOF (Note: the Code does not apply to privately managed forest lands). The ministry official with the power to accept or reject operational plans submitted by forest companies is called the District Manager.

Unfortunately, the *FPCBC Act* and regulations provide District Managers with broad discretionary powers in setting the terms and conditions under which forestry companies may operate in and around streams, regardless of their fish bearing status. For example, s. 73(1) of the Operational Planning Regulation (B.C. Reg. 174/95) specifies the minimum riparian reserve zone widths and riparian management zone widths to be left along streams of various classes, while s. 73(3) allows the District Manager to vary these widths. Additionally, s. 8(1) of the Timber Harvesting Practices Regulation (B.C. Reg. 181/95) states that, *"a person carrying out a timber harvesting operation on applicable land must not yard or skid timber through or over any stream or fisheries-sensitive zone unless yarding or skidding is authorized in a logging plan."*

All of the provisions of the FPCBCA and regulations that apply to timber harvesting adjacent to streams contain this discretionary wording, which, if applied to operational plans being submitted by Licensees, would make it virtually impossible to hold forest companies accountable should their actions result in damage to fish habitat. Furthermore, it would be extremely difficult to hold a District Manager, who approves operational plans while exercising his/her discretion in a manner that could favor the forest companies, accountable because the penalty provisions of the FPCBCA do not apply to government (s. 1(5)).

Another problem with relying on the Forest Practices Code to protect fish habitat from the ravages of logging, lies with the definition of a fish stream within the Code itself. The Code defines a fish stream as that portion of a stream that is frequented by fish. This means that for the provisions of the Code dealing with riparian protection (Table 1) to take effect in a situation involving a proposed clearcut next to a fish bearing stream; the Licensee need only consider whether that portion, or reach, of the stream immediately within or adjacent to the proposed cutblock is frequented by fish. It does not take into account downstream fish values. We believe this is a very dangerous and short-sighted view of what constitutes fish habitat. Under this scenario, the Code permits the removal of all timber adjacent to non-fish bearing stream reaches. It also specifically permits cross-stream falling and yarding across these sections of stream. Cross-stream yarding is even defined in the Riparian Management Area Guidebook (B.C. Ministry of Forests and B.C. Ministry of Environment, Lands and Parks 1995) as: *"...when a log or portion of a log is yarded over a stream and contacts any part of the bank, channel or vegetation in the RMA."* This is a recipe for disaster in that cross-stream yarding is one of the most destructive activities associated with timber harvesting leading to streambank destabilization and the introduction of logging debris into streams.

Given that most of the non-fish bearing reaches of fish producing streams on the B.C. coast are situated in the steeper headwater areas, where the requirements for riparian management areas (RMAs) are less restrictive, and where cross-stream falling and yarding would prevail as an acceptable forest practice, it is almost certain that fish habitat will continue to be lost as the effects of these land-use practices are telegraphed to fish-bearing stream habitats at points located further downstream.

Additionally, following proclamation of the Forest Practice Code on June 15 1995, the government in British Columbia immediately began relaxing the stated Code requirements with regard to the percentage of timber that must be left standing within RMAs (Table 1). This is being done to fulfill a government promise to the forest industry that any reduction in the Annual Allowable Cut (AAC), that may arise as a result of strict application of the Forest Practices Code requirements concerning the amount of timber to be left in reserve areas, would not exceed 6%.

The Fisheries Act

In British Columbia then, with the exception of certain administrative penalties under the FPCBCA, regulatory protection of the fishery resources from possible damage resulting from forest harvesting activities lies solely within the federal *Fisheries Act*. This is recognized in Sections 26(1), 35(1) and 36(3) of the Act, which read as follows:

Section 26(1)

> "One-third of the width of any river or stream and not less than two-thirds of the width of the main channel at low tide in every tidal stream shall always be left open, and no kind of net or other fishing apparatus, logs or any material of any kind shall be used or placed therein."

This section can apply to log jams caused by inadequate cleanup, or encroachment of bridges and road building material (side cast) on stream channels.

Section 35(1)

> "No person shall carry on any work or undertaking that results in the harmful alteration, disruption or destruction of fish habitat."

Note: the key word in this section is harmful.

This section can apply to the following situations; removal of all riparian (stream side) vegetation; falling and yarding of logs across, in and through (not over) streams; driving heavy equipment in and through streams; landslides and debris torrents which are directly attributable to logging-related activity; prolonged sedimentation of streams caused by poor road construction practices and/or maintenance; and culvert failures resulting from inadequate maintenance.

Section 36(3)

> "... no person shall deposit or permit the deposit of a deleterious substance of any type in water frequented by fish or in any place under any conditions where the deleterious substance or any other deleterious substance that results from the deposit of the deleterious substance may enter any such water."

This section can be used when sediment is discharged into streams from streambank erosion, from uncontrolled road or ditch erosion, and also when there is a discharge of gasoline, oil, or grease from logging equipment and/or spills.

Enforcement of the *Forest Practices Code of British Columbia Act* and the Federal *Fisheries Act*

The Forest Practice Code of British Columbia Act

Enforcement of the FPCBCA rests with the MOF and the MOELP.

The FPCBCA was only proclaimed in effect on June 15 1995, therefore, it is too early to assess whether compliance and enforcement under the terms of the Act will have any direct, positive, long-term impact in terms of protecting fish habitat. However, our analysis of the provisions of the Code concerning stream protection measures leads us to believe that little will change unless there is a concerted effort on the part of Licensees and the government approving bodies to comply with the stated spirit and intent of the Code respecting the conservation of "*...biological diversity, soil, water, fish, wildlife, scenic diversity and other forest resources,*" and, "*restoring damaged ecologies.*" Thus far, however, we have found that planning and implementation of timber harvesting strategies in some regions of the province is proceeding as if the Code was not yet in effect (Sierra Legal Defence Fund and Forest Policy Watch 1996). This, for the most part, is because the Code is being phased in gradually over a 2-year period.

The federal Fisheries Act

Under the *British North America Act*, the federal government has jurisdiction over both coastal and inland fisheries. However, in British Columbia, responsibility for management and protection of resident sport fish, including steelhead and coastal cutthroat trout, rests with the province and falls within the jurisdiction of the B.C. MOELP. The federal Department of Fisheries and Oceans has retained responsibility for marine and other anadromous fishes. This includes the five species of Pacific salmon, herring, and shellfish. Thus, in British Columbia, at least in practice, the *Fisheries Act* is administered by both federal Fisheries Officers, who are responsible for all waters where anadromous salmonids are found, and by provincial Conservation Officers in areas where the streams contain sport fish. This division of responsibility between the two governments is a very loose arrangement and is the result of a written policy agreement, but it is not legislated.

Table 1. Specified minimum Riparian Management Area (RMA) slope distances for riparian stream classes as set out in the *Forest Practices Code of British Columbia Act*

Riparian class	Average channel width (m)	Reserves zone width (m)	Management zone width (m)	Total RMA width (m)
S1 large rivers	≥100	0	100	100
S1 except large rivers	>20	50	20	70
S2	>5≤20	30	20	50
S3	1.5≤5	20	20	40
S4	<1.5	0	30	30
S5	>3	0	30	30
S6	≤3	0	20	20

Note: All tabled results for S1 through S4 occur within fish streams and community watersheds; tabled results for S5 and S6 exclude fish streams and community watersheds.
Source: Forest Practices Code Riparian Management Area Guidebook, December 1995.

The only provincial environmental enforcement organization in British Columbia is the 140-member Conservation Officer Service, which is part of the provincial MOELP. However, about 80% of their time is spent enforcing the hunting and fishing game laws, parks protection, wildlife control, public education, and administration (Doug Turner, Senior Enforcement Officer, B.C. MOELP, Vancouver Island Region, Nanaimo, B.C., personal communication). Only 20% of their work involves enforcing the *Waste Management Act*, the *Water Act*, the *Pesticide Control Act*, and the *Fisheries Act* as it relates to fish habitat.

The DFO is responsible for reviewing all logging operations that may impact upon streams containing anadromous fish, but this agency, too, is lacking the manpower to properly enforce the provisions of the Fisheries Act. Often, the same individual officers who are responsible for enforcement, are those that approve the initial logging prescriptions for the companies.

Considering the immense geographic size of the province (about 1 100 000 km^2) and the literally hundreds of thousands of kilometres of coastal and inland waters, both of these enforcement agencies are sadly lacking the staff required to effectively enforce the *Fisheries Act* as it relates to the logging industry on a province-wide basis.

Over the last 3 years that we have been conducting independent investigations into fish habitat damage caused by logging, we have discovered that the government enforcement agencies often lack expert scientific and legal advice, direction, and support from within their agencies. Recently, our investigators were asked for both scientific and legal advice by a DFO Fisheries Officer who was conducting an investigation into fish habitat damage and who indicated that such advice and support was not available from within the government. It has also become quite clear to us that both the provincial and federal enforcement agencies are very restricted in their enforcement activities because of the close relationship between government officials and the logging industry.

In 1989, a DFO official, Mr. Otto Langer, then head of the Habitat Management Unit for the DFO in British Columbia, wrote a memo to his superiors that was subsequently leaked to the press. This memo documented many instances of high levels of government interfering with the proper enforcement of the *Fisheries Act* because of the "negotiate and compromise at all costs philosophy" of both the provincial and federal governments.

Langer said that a special relationship existed between governments and large corporations with

the result that their violations of the *Fisheries Act* were often never prosecuted. He further suggested that only small companies and individual citizens were being prosecuted under the Act and that, "...it must be appreciated that DFO habitat enforcement has reached an all-time high in inconsistency and an all-time low in terms of the adverse precedents they are setting and the habitat destruction that has taken place is being ignored to a significant degree."

In dealing with government officials over the last year, we have found that little has changed since this memo was publicly released. Quite often there appears to be an incestuous relationship in existence between both federal and provincial government officials and the logging companies. More than one official from the MOF has referred to the logging companies as their clients and this relationship results in thousands of logging approvals simply being routinely granted on most of the applications to clear-cut, usually without any on-site government inspection.

At the present time there is virtually no active enforcement policy in B.C. for *Fisheries Act* violations relating to improper logging practices. In fact, the opposite is true. Our investigators have found that government officials within the DFO and MOELP openly express a negative attitude toward the concept of enforcement through prosecution under the *Fisheries Act*. Time and again, these officials have indicated that cooperative regulation of the forest industry is the answer. In fact, we have since found out that this is government policy at both the federal and provincial government levels.

The problem with this kind of policy is that it sets the stage for a very weak, but possible shield against prosecution, that of "officially induced error" When the government regulatory agencies become involved in the planning process as it is practiced in regards to timber harvesting in B.C., it sets in motion a procedure of review and approval of logging plans wherein the government (MOF, MOELP, and DFO) has an opportunity to dictate the necessary steps, or prescriptions, that should be followed by a licensee to ensure fish habitat is protected. Often, the reviews of the proposed plans are superficial and/or the fisheries resource data (i.e., fish presence/absence, results of fish habitat inventories, etc.), if any, upon which these reviews are based are inadequate or incorrect (personal observation). The resulting prescriptions themselves are often inadequate or incorrect. However, the licensees are obliged to follow these plans and prescriptions as specified either under provisions of the applicable legislated statute, such as the FPCBCA or the *Fisheries Act*, or under the terms and conditions of their Cutting Permits or Licenses to Cut.

Under this scenario, when, or if, damage to fish habitat occurs as a result of the Licensee's on the ground activities, whether they are in compliance with their permit or not, charges against the offenders are seldom considered, or, if they are, they are seldom pursued by the Crown. This is because the current mind set in B.C. of both the provincial Attorney General's office and the federal Department of Justice is that if a corporation is operating under a plan, permit, or license approved by government, and their works or undertakings lead to damage to fish habitat, then the government, not the Licensee is to blame because it permitted the act that led to the offence (P. Ewert, Criminal Justice Branch, B.C. Ministry of Attorney General, personal communication). This is particularly so if the corporation is violating its approved plan, permit, or license and the government enters into lengthy arbitration with the corporation while the offence is ongoing (tacit approval) in an effort to rectify the situation through negotiation and compromise. This occurs even though virtually every permit or license issued in the province contains wording to the effect that adherence to the terms and conditions of the approval document does not absolve the Licensee or Permittee from their obligations under other existing provincial or federal statutes, such as the *B.C. Waste Management Act* or the federal *Fisheries Act*.

Crown prosecutors also apply a test of reasonableness to all charges brought before them to determine whether the alleged offender was duly diligent. Due diligence applies if a person has taken all reasonable measures that a person should reasonably take to prevent an offence from occurring. This could be, and often is, interpreted to mean simply following the conditions set out in the permit. Due diligence is the only defense available under the *Fisheries Act*.

Finally, the provincial Attorney General also considers whether it is in the public interest to proceed with a charge as part of their charge approval standard. In several instances, the Attorney General's office has refused to proceed with charges against an offender because during the period between commission of the offence and the laying of charges, the offender may have, either voluntarily or as directed by government regulatory agencies, participated in efforts to rectify the damage done. In such cases, it would be determined that it is not in the public interest to proceed with charges because the offender

cooperated in efforts to clean up their mess. Also, government agencies, such as the MOF, who also conduct logging through their small business forest enterprise program, often escape prosecution because the Attorney General's office has adopted the position that it is not in the public interest to proceed with charges against a government agency since the taxpayer would have to foot the bill for any fines levied against the agency charged (P. Ewert, Criminal Justice Branch, B.C. Ministry of Attorney General, personal communication).

The net effect of applying the combined tests of officially induced error, due diligence and whether it is in the public interest to proceed, is that very few charges are ever laid against logging companies or the MOF for damage to fish habitat caused by their negligence.

An alternative model—enforcement in Ontario

In the central Canadian province of Ontario, environmental prosecutions received very little emphasis prior to 1985. Regulators had concentrated on administrative remedies similar to those in B.C., almost to the exclusion of prosecutions. They have since found this to be ineffectual. In 1985, there was a major change in Ontario's environmental policies, which led to the creation of a special branch of environmental police, an increase in prosecutions staff, and the introduction of severe penalties for polluters (personal observation). By 1989, the number of prosecutions had increased 500%, and fines levied for those offences increased dramatically. These prosecutions received much media coverage and many experienced environmental experts agree that this emphasis on prosecutions had a profound effect on improving the environmental compliance of many corporations.

The Salmon Habitat Protection Project

In September 1992, the Sierra Legal Defence Fund introduced a new initiative to protect our remaining salmon habitat from destructive logging practices. We have been working with a coalition of private citizens, public interest groups (Sierra Club of Western Canada, Steelhead Society of B.C., United Fishermen and Allied Worker's Union, Greenpeace, Friend's of Clayoquot Sound), and aboriginal peoples to locate and investigate sites where fish habitat has been damaged by logging. We then bring this information to the government's attention and, where the government fails to act, began to initiate private prosecutions under the *Fisheries Act*. This initiative has been expanded to include the monitoring of Licensee compliance and government and Licensee compliance and enforcement under the recently proclaimed *Forest Practices Code of British Columbia Act*.

A fisheries biologist and a soils scientist have been retained to work with our clients, to conduct field investigations of logging-related stream damage and to document evidence for potential cases. The scientists work in concert with a lawyer who assists them in the legal aspects of conducting an environmental investigation and in the preparation of prosecution briefs, where necessary.

Private Prosecutions

Although in the United States the private citizen has virtually no formal role to play in the criminal justice process, the Canadian criminal justice system has long recognized the importance of the right of a citizen to lay a criminal charge and to conduct a private prosecution. This right has been codified in the Canadian *Criminal Code*, and it is rooted in the English common law where historically, before 1555, all criminal prosecutions were of a private nature. This right to conduct private prosecutions continued in England and had become a tradition by the time Canada enacted its *Criminal Code* in 1893, which was really a statutory restatement of the traditional English common law.

In Canada, when governments refuse to act, private prosecutions provide an opportunity for everyone to become involved in bringing violators before the criminal courts to face conviction and punishment. Citizen action can therefore be extremely powerful and effective in deterring corporations from committing crimes against the environment.

Private prosecutions have always played an important role in Canada and the Canadian government has even encouraged private prosecutions under the *Fisheries Act* by enacting the *Penalties and Forfeitures Proceeds Regulations*, which provide that upon conviction of an offence under the *Fisheries Act*, as a reward, one-half of the fine shall be paid to the individual citizen who commenced the private prosecution. Normally, provincial or federal government prosecutors conduct environmental prosecutions in Canada, and the government can intervene in a private prosecution and stop it, or take over and prosecute it.

If sufficient damning evidence has been gathered by a citizen, government officials may, for political reasons, be reluctant to stop the prosecution. However, it has been our experience in British Columbia that recent attempts at launching private

prosecutions using the federal *Fisheries Act* have been thwarted by government intervention in these cases.

The Sierra Legal Defence Fund has, over the past 3 years, initiated four private prosecutions in B.C. on behalf of public interest groups: two against the Greater Vancouver Regional District (GVRD) and the Greater Vancouver Sewerage and Drainage District (GVS&DD) relating to the illegal discharge of sewage into fish-bearing waters; and two against logging companies relating to a logging-related damage to fish habitat. The Crown, in the guise of the Attorney General of B.C., has intervened in all four cases. Three of the cases have been stayed (*R. ex rel Werring v. GVRD and GVS&DD, R. v. Fletcher Challenge Canada Limited, Crown Forest Industries Limited and John Kay,* and *R. ex rel Werring v. Coast Mountain Hardwoods Incorporated, Her Majesty the Queen in Right of the Province of British Columbia* (as represented by the Minister of Forests) *and Bob Brash*). The third *(R. ex rel Werring v. GVRD and GVS&DD)* is still before the courts. A stay of proceedings means the charges have been held in abeyance, either pending further review, or because the Crown does not intend to proceed with the charges. In the aforementioned cases, where a stay of proceedings was entered, the Crown determined it would not proceed with the charges. Following a stay, the charges can be reinstated within 1 year; however, this is seldom done.

In a stay of proceedings, the Crown is not obliged to give reasons for their decision. However, we were told by the Crown that, in the case of the stay of charges against the GVRD and the GVS&DD, they felt it was not in the public interest to proceed. Officially, the Crown announced publicly that, despite impeccable evidence supporting our case, it was felt that the Crown would not be able to achieve a conviction because of a hand-shake agreement between the polluter and the regulatory agency, the B.C. MOELP.

In regards to the charges against Fletcher Challenge and Crown Forest, we were informed by the Crown that the case was stayed because they could not find a government expert who would provide supportive evidence to uphold the charges. In fact, the expert the Crown relied on later informed us that he did not want to be involved in the case for personal reasons that could lead to a conflict of interest, that he felt he lacked the expertise to provide the assessment the Crown was seeking, and that he informed the Crown prosecutor of these matters but he was obliged to provide an opinion anyway. No attempt was made by the Crown to seek another opinion.

In the case of the charges against Coast Mountain, the government and a former MOF District Manager, we were told by the Crown that they accepted MOF's explanation that the road construction that led to the alleged damage to fish habitat took place over an existing road (the existing road was actually a decades-old skid trail that had been upgraded to a modern road) and that it was not in the public interest to proceed because there was "... a reported improvement which was assisted by remediation activity conducted by the logging company under the guidance of a federal Fisheries Officer." (letter to SLDF, dated October 28, 1997, from K.W. Ball, DuMoulin and Black, Barristers and Solicitors, Independent prosecutor retained by the Attorney General of B.C. to handle the case on behalf of the Crown).

Finally, we have been told by the Crown that the B.C. Attorney General has a policy of intervening in all private prosecutions brought forth in this province. Additionally, the current Minister of Environment is on record as saying that he does not support the idea of private citizens laying charges against corporations.

Conducting On-site Investigations

Since September 1993, our investigators have traveled over 16 000 km (10 000 miles) in this vast province in response to reports from our clients that damage to fish habitat has occurred from logging and logging-related activities. There are inherent difficulties in reaching these often remote sites, some of which include: long, arduous travel over rough logging roads; access that may be restricted to boat, float plane, or helicopter; and, the necessity to return to the site three or four times because of weather conditions or to acquire other expert opinions, which are always necessary when putting together a case.

The preliminary investigation entails conducting detailed measurements of the stream and its fish habitat characteristics, fish sampling, documenting the extent of the damage, if any, and taking detailed notes, sketches, photographs, and video footage of the site. This may require several days of work.

Following the preliminary on-site visit, our investigators then visit the area-specific offices of the MOF and MOELP to search for, locate, and copy all relevant documentation pertaining to a particular case. These documents are carefully scrutinized and

catalogued during the initial search to ensure that all requested documentation is forwarded to us by the respective ministries. In some cases during this process, we have found important documents that were not provided to us.

We have also found that the government agencies attempt to thwart our requests for information by levying exorbitant charges for document retrieval. For example, a recent search for one case resulted in the MOF billing our organization over $600.00 to obtain photocopies of the relevant documents. However, these fees were avoided by invoking the argument that the search was being done in the public interest. Public interest exemptions for costs in these cases are usually granted.

To locate active logging sites our investigators visit regional and district Forestry Offices and review logging plans and maps and talk to MOF staff. We have found, however, that when we do this, the MOF often warns the logging companies that we are in the area investigating them.

Workshops

Throughout the term of this project, our investigators visited several logged sites where independent third parties had reported observing logging-related damage to fish habitat, but when our investigators arrived on-site, they could not find any fish habitat damage that was directly attributable to logging. Although all of the areas visited in these instances had some logging occurring adjacent to streams, the streams themselves, for the most part, had remained intact. It became apparent that the informants had observed and mis-construed as "damage to fish habitat" the following:

- natural windfall or deadfall trees lying in and/or across streams;
- naturally induced landslides which have deposited debris in streams; or,
- where instances of habitat damage did occur, it was in sections of streams or streams that did not support fish and, therefore, did not constitute damage to fish habitat.

Unfortunately, a great deal of time and money was spent investigating these sites.

Responding to reports from "uninformed" sources can divert the efforts of our investigators from important tasks and rapidly drain travel budgets. In addition, situations could arise where private citizens might report inaccurate or incomplete information to government enforcement agencies and thereby damage their credibility when it comes to trying to invoke a government response in other, valid situations.

To combat this dilemma the Sierra Legal Defence Fund developed an information workshop dealing with issues related to logging and fish habitat damage. The workshop is designed to provide participants with an understanding of what does or does not constitute damage to fish habitat caused by logging. It covers topics such as:

- what is fish habitat;
- the *Fisheries Act* and how it can be applied to logging;
- the B.C. *Wildlife Act* and how it can be applied to logging;
- interpreting and applying the *Forest Practices Code of British Columbia Act*;
- how to locate and identify areas where logging is taking place or about to take place;
- how to obtain evidence of a prosecutable offence being committed;
- when and who to report to;
- the use of notes, water quality sampling, and video and still cameras for obtaining evidence; and,
- preserving and maintaining continuity of evidence.

The objective of these workshops is to train people in the art of detecting and reporting potential violations of both the *Fisheries Act* and the *FPCBC Act* so that the information provided by an informant is as accurate and complete as possible. In addition, the workshops serve to increase the number of informed observers in the field watching over logging operations, thereby forcing the logging companies to act responsibly. In this way, limited financial resources can be maximized by ensuring that only those complaints that prove to be valid are investigated.

Our first workshop, hosted by the Western Canada Wilderness Committee, was held on October 12, 1993. Twenty-two people from various environmental organizations (Western Canada Wilderness Committee, Vancouver Temperate Rainforest Action Coalition, United Fishermen and Allied Worker's Union, Canadian University Students Overseas - CUSO, Sierra Club of Western Canada, Van City Credit Union) and universities and colleges (University of B.C., Simon Fraser University, B.C. Institute of Technology), and members of the general public attended the event. We have since conducted

over 50 other workshops in various regions of the province.

These and other workshops, as they arise, should be instrumental in raising public awareness on the logging and fish habitat conflict throughout B.C. By bringing this issue to the forefront in as many areas as possible, we hope to achieve our objective of protecting fish habitat in all regions of the province rather than in a few isolated areas.

Project Findings

The Sierra Legal Defence Fund has investigated public complaints concerning logging-related damage to fish habitat in over 60 different watersheds in the province over the past 5 years (Sierra Legal Defence Fund 1993, unpublished report; Sierra Legal Defence Fund 1994, 1995, 1996, 1997). Our findings clearly show that there are a number of problems that are common to most of the sites where fish habitat damage has been documented, some of which are:

1) Streams are being misclassified.

   Often the streams draining logged and proposed cutblocks are small and seemingly insignificant. However, they often provide important spawning and/or rearing habitats for resident fish or for fish from larger systems to which they are tributary. These smaller streams are frequently classified as non-fish bearing streams. This practice, whether intentional or not, enables logging companies to operate under less stringent guidelines that can result in significant stream damage. As a consequence, valuable habitat is still being lost.

2) Poor road building and maintenance is resulting in sediments from road and ditch erosion and landslides entering and damaging fish habitat.

   Companies sometimes do not follow accepted road building guidelines that are designed to prevent erosion and protect streams from sediment contamination.

3) Inadequate leave strips are being left along important fish bearing streams.

   Leave strips protect streams by providing a buffer zone, which keeps logging activity away from the streams. However, in the past the B.C. Fisheries/Forestry Guidelines and more recently the Forest Practices Code of British Columbia specify that leave strips should be wider and more wind-firm than what we are seeing in the field (personal observation). The leave strips we typically see are too narrow and they inevitably blow down causing massive damage to streambanks and downstream areas (by forming huge log jams). The companies often argue that approval to log these cutblocks was given years ago, prior to implementation of the current guidelines and, as such, they are not bound to follow them.

4) Resource agencies (DFO and MOELP) often approve the clear-cutting of, and road building adjacent to, fish-bearing streams.

   The DFO and MOELP typically only provide input on the logging of a cutblock at the 5-year development plan level. Often, there is insufficient information provided at this level of planning for these agencies to make a proper, informed decision whether or not to allow logging, or request modifications to the proposed logging plans. Funding and staffing constraints prevent them from checking each and every cutblock and to speed things up approvals are granted without specific conditions attached for the proper protection of fish habitat.

5) Logging companies do not always follow approved logging plans.

   In some cases, DFO and/or MOELP make specific recommendations on how a cutblock should be logged to best protect fish habitat. This is usually done following a field review. In some cases, however, the companies are not following these recommendations, with disastrous consequences for fish streams. Often DFO and MOELP are not informed when this occurs, but even if they find out about it, they are reluctant to prosecute because of their special relationship with industry.

6) The DFO and/or MOELP often determine that the degree of habitat loss, where it does occur, is too insignificant to be concerned with.

   We have documented several cases where the amount of habitat loss is on the order of several hundred square metres, or where logging debris deposited in a stream has formed a complete blockage to upstream or downstream migration (Sierra Legal Defence Fund 1994). However we have also found that DFO and/or MOELP officials usually consider this to be an insignificant

impact and as such will not even take the time to investigate the cause of the habitat loss or request that it be cleaned up.

7) Once a cut-block containing fish-bearing streams, or lying immediately adjacent to a fish-bearing stream, has been approved for logging and logging takes place, there is little or no follow-up by either DFO or MOELP officials to see whether specified prescriptions for the protection of fish habitat have been followed.

   We have found that this is due to the lack of available staff in both regulatory agencies. For the most part, the habitat protection staff in both agencies are office bound, reviewing and approving proposed timber harvesting plans, while the understaffed enforcement units for both agencies are engaged in other activities, such as wildlife control (e.g., errant bears), apprehending poachers, and/or monitoring the commercial and recreational fisheries. However, several provincial conservation officers have told us, confidentially, that they have been directed to stay away from the timber industry and to let them police themselves.

8) Jurisdictional overlapping between the provincial and federal authorities concerning administration of the *Fisheries Act* results in many offenders going unpunished.

   In some cases, DFO staff become involved in the planning of the logging of a cut-block adjacent to fish-bearing waters to the exclusion of MOELP. However, there have been cases where specific prescriptions made by DFO for the protection of fish habitat were not followed, resulting in subsequent habitat loss in a section of stream that is found to support only resident sport fish. In these cases, there is a tendency for the DFO to opt out of the situation and pass it on to the MOELP who had little or no involvement in the planning of the block in the first place.

Project Evaluation

The overall goal of this project was to strengthen government enforcement of the federal *Fisheries Act* and the new FPCBCA as they relate to timber harvesting and fish habitat loss. On the whole, we are pleased to report that the project has met with some success. Public recognition of the problems surrounding lack of enforcement has increased over the last 3 years. Governments' attitude towards enforcement has changed dramatically. Enforcement activity has increased somewhat and, *most importantly*, the forest industry now perceives enforcement as a realistic possibility and a real deterrent. However, we believe there is still a large segment of the provincial bureaucracy that believes the idea of tough enforcement is dubious, and there is still a long way to go before the forest industry views government enforcement as a real threat to their way of doing business.

Government enforcement activity has increased somewhat over the life of the project. In 1992/93, the first year of the project, no charges had been laid against any forest companies for destruction of fish habitat, even though our investigators found that damage to fish habitat was pervasive throughout the province (Sierra Legal Defence Fund 1994). Only four investigations of suspected offences were conducted. By 1993/94, this had increased to eight prosecutions arising out of almost 40 investigations. For 1994/95 the B.C. government reports that over 70 charges were laid. There were 27 convictions recorded for forestry-related offences (however, several of these were charges laid against small, independent operators, not large corporations) and 329 investigations into potential or suspected environmental crimes relating to timber harvesting in this province.

We consider this to be a more acceptable level of enforcement compared to previous years. However, there is still a long way to go before the enforcement level reaches a point where a realistic deterrent to logging companies is provided that would forsake fish habitat for corporate profits. Recent studies conducted by Sierra Legal's forestry team clearly show that despite the introduction of the Code and despite the promises of stiff fines and tough enforcement, fish habitat in this province is still being lost to the ravages of logging, with the full knowledge and consent of the government regulatory agencies (Sierra Legal Defence Fund 1996, 1997). Sierra Legal's findings concerning fish stream destruction (Sierra Legal Defence Fund 1997) were confirmed by a joint MOF/MOELP review team (B.C. Ministry of Forests and B.C. Ministry of Environment, Lands and Parks 1997).

For the most part, most of the fines levied against corporations for environmental offences under the *Forest Practices Code of British Columbia Act*, to-date, are administrative penalties amounting to a few thousand dollars. Further, as of November 1997, (more than 2 years after the Code was proclaimed) there has not been one single charge laid by government against a logging company under the environmental protection provisions of the Code (J. Hunter,

Compliance and Enforcement Branch, MOF, personal communication). Not even in relation to documented damage to fish streams by the government's own review team (B.C. Ministry of Forests and B.C. Ministry of Environment, Lands and Parks 1997). Only three charges have been laid under the Code and they relate to unauthorized timber harvesting (cutting outside approved boundaries). The Sierra Legal Defence Fund, on the other hand, laid the first charges under the environmental protection provisions of the Code on July 15, 1997 (*R. ex rel Werring v. Coast Mountain Hardwoods Incorporated, Her Majesty the Queen in Right of the Province of British Columbia* (as represented by the Minister of Forests) *and Bob Brash*) on behalf of public interest groups and First Nations, but the provincial Attorney General intervened and stayed the charges.

Therefore, despite government promises of tough enforcement and the specter of $1 000 000 fines for offences leading to environmental damage, logging companies continue to cause damage to fish habitat and the FPCBCA remains an enigma as to whether it will result in significant changes in the way forestry is to be conducted in this province. Certainly, while the hammer is there to force change, it is not much of a deterrent if government is unwilling to wield it.

Despite these shortcomings, we believe that this project was a likely factor (and perhaps the major factor) in the improvement in forestry-related enforcement in this province. The increased enforcement activity at the provincial government level has come as a result of clear and strong direction from the top political levels, primarily from the Minister of Environment. The Minister has made strong public statements about the need for improved enforcement, which has percolated down to improved line staffing for enforcement activities. The Minister has credited Sierra Legal Defence Fund's activities publicly, and assured us privately, of this project's direct impact on the government's newfound direction. It remains to be seen, however, whether this change in attitude is infectious and whether it will spread throughout government, particularly to the MOF.

Perhaps the most significant evidence that a new government attitude is in the offing however, are the slogans adopted by the B.C. government to advertise its new FPCBCA, such as: "Changing the way we manage our forests" and "Tough enforcement." These slogans are featured on most code-related material and have continued to be the message of the government's recent media releases. In addition, the forest industry is also advertising slogans such as: "A new day in the forest" and "The most stringent environmental laws in the world."

Polling results show tremendous support for this strategy. A Viewpoints Research poll conducted in 1994 showed 85.7% of the B.C. public believed "heavier penalties and tougher enforcement of better forest practices are long overdue in B.C." Significantly, this concern has not slackened with the introduction of the new code. An Angus Reid poll conducted in March 1996 included a question regarding opinions of current government penalties against companies whose practices are "harmful to the environment." An amazing 80% felt penalties should be tougher (28%) or a lot tougher (52%) and only 3% thought they should be relaxed.

Summary

From our work (Sierra Legal Defence Fund 1994, 1995, 1996, 1997) and the work of others (Tripp et al. 1992; Tripp et al. 1993; Tripp and Grant 1993; Tripp 1994; Northwest Hydraulic Consultants 1994), it is apparent that vast quantities of fish habitat in B.C. have been devastated as a result of timber harvesting and its associated activities such as road building, both past and present. Much of this habitat loss has and is occurring with the full knowledge and approval of both provincial and federal government officials. We can only surmise that this is because of the historical relationship that has developed between government agencies and the logging industry. This special relationship has resulted in a negotiate and compromise attitude and a resultant lack of enforcement of environmental statutes, such as the *Fisheries Act*, as they relate to negligent logging practices.

The government of B.C. has finally lived up to its promise to enact new legislation governing timber harvesting in this province with the proclamation of the FPCBCA on June 15 1995. The Code provides for fairly stiff penalties for offenders, which include fines that can be levied administratively ($1000–$50 000) or by the courts ($5000–$1 000 000), and/or imprisonment (6 months to 3 years). However, it remains to be seen whether the political will exists to enforce compliance with the new Act. We are not convinced that this is so, given the track record of the provincial government over the past 50 years. Additionally, we are not convinced that there is yet in place an adequately staffed environmental police force that is completely independent of bureaucratic or political influence.

Traditionally, both the B.C. and the federal government regulatory agencies have adopted a "negotiate and compromise at all costs" position in dealing with the forest industry in B.C. It is not working. Fish habitat is still being compromised. It is for this reason we have adopted a "get tough with offenders" approach. This approach is working. Since we undertook this initiative, there has been a significant shift in government policy and attitudes towards the protection of fish habitat from the ravages of timber harvesting and other land management practices affecting aquatic ecosystems in this province. This has been expressed in more charges than ever being laid against forest companies who violate the law, particularly where fish habitat is a concern.

We propose to continue our investigations and to bring our findings to the attention of the public, existing enforcement agencies, and both levels of government (provincial and federal) to pressure them into establishing a clear and workable environmental enforcement policy and adequate environmental police forces to protect fish habitat.

We strongly feel that our continued involvement in conducting our own private investigations will assist in bringing about these changes.

Many of the opinions expressed in this paper are those of Sierra Legal Defence Fund, and these views may not be shared by other stakeholder groups.

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The Recovery of Degraded Riparian Systems with Improved Livestock Management

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Abstract

Historic and current livestock grazing practices have created severe impacts upon the natural resources of most public and private rangelands in the western United States. These impacts result in eroded uplands and stream courses, lowered water tables resulting in wet meadows becoming dry sagebrush flats, degraded water quality, increased flood events with a reduction of late-season flows, degraded fish and wildlife habitat, lost recreation opportunities, and degraded visual quality. With improved livestock grazing practices, dramatic improvements in rangeland and riparian area condition can be achieved. This improvement results in a restoration of many of the impaired resource conditions and improvement of recreation opportunities. As the District Ranger for the Twin Falls Ranger District of the Sawtooth National Forest, the author's efforts to gain compliance from livestock grazing permittees resulted in an agreement by Forest Service officials in 1989 to transfer him out of the area. The author filed a whistleblower complaint with the Secretary of Agriculture's Inspector General, and as a result of the investigation, he was completely exonerated and has continued to restore the resources on his district through improved livestock grazing practices. The Twin Falls Ranger District has implemented a number of very successful demonstration exclosures and riparian pastures, which dramatically show the ability of rangelands and riparian areas to recover from 130 years of livestock grazing impacts.

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Introduction

The Twin Falls Ranger District of the Sawtooth National Forest has made diligent efforts since the fall of 1986 to improve riparian areas and upland rangelands, which have been degraded by 130 years of inappropriate livestock grazing practices. As District Ranger, I have insisted that the 34 livestock grazing permittees, who now graze cattle or sheep on the District, comply with the terms of their permits and associated guidelines designed to improve the natural resources and recreation opportunities of the area.

The impacts upon the resources from past and current livestock grazing practices, as my staff and I observed them in 1986 and 1987, were obvious. It was also obvious that the continued degradation could not continue if the remaining resource values were to be preserved for future generations. We also knew that the resources could improve from their present condition with improved management; watershed conditions and fish and wildlife habitat, especially, could be greatly improved. We were determined to reverse the trend of most resources being degraded, to a course of improving them for future generations. These improved conditions would also serve to help keep the livestock permittees in business by improving livestock forage and by eliminating concerns from the public, who own the national forest lands.

We felt that we had the direction to make the needed changes. It was national policy that the agency make the changes necessary to improve rangeland and riparian area conditions as we graze livestock. However, when we began to require the needed changes in management of the permittees (voluntary improvements were the rare exception), many of them resisted vigorously. They were able to muster help in preventing any changes to their traditional methods of operation from within the Forest Service and from their political representatives. Through continuing to work with the permittees, and resisting inappropriate outside and agency actions, we were able to make significant improvements in management of the 18 grazing allotments on the District.

We have implemented numerous riparian area improvement projects. These include fenced riparian pastures and a number of exclosures where grazing is no longer allowed. These demonstration projects have been invaluable as examples of the ability of the riparian areas and uplands to recover from past and present livestock impacts. These demonstration projects have also helped make it possible to gain significant improvement of most riparian areas and rangelands across the rest of the District.

My framework of discussion is to present some of the history of livestock grazing on the Twin Falls Ranger District, a detailed description of the resource problems, how we were able to achieve needed management changes in the face of great opposition, and the resultant improvement in condition of the riparian areas and uplands. I also describe the Dry Gulch riparian improvement project in some detail as one example of the many projects we have implemented.

I was also asked to discuss the controversy that developed in our grazing program in 1989, which received extensive national media coverage. That is the only reason our work has received enough attention that the coordinators of this conference heard about it. I have covered it briefly, because it is a very long story.

Study Area

The Sawtooth National Forest in south central Idaho includes four ranger districts and the Sawtooth National Recreation Area. The Twin Falls Ranger District is a gentle, isolated mountain range adjacent to the Nevada border. It is a primary recreation area for many of the 100 000 people who reside in the Snake River Valley near the city of Twin Falls. It has one of the largest grazing programs of all the districts in the 16 national forests of the Intermountain Region of the U.S. Forest Service.

Background

I came to the District in the fall of 1986, as did my Supervisory Rangeland Management Specialist Ralph Jenkins. We found livestock grazing operations that were destructive and in violation of the terms of the grazing permits. The obvious problems included very heavily grazed uplands and riparian areas; streams severely trampled and eroded with greatly reduced water tables; large dry gullies eroding deeper with every storm; salt placed near water, further concentrating cattle; cattle in every unit of most allotments, essentially creating season-long grazing instead of a rotation system; cattle on some allotments long after the off-date in the fall; fences and water developments (spring head boxes, pipelines, troughs, and ponds) in disrepair; sheep bedding on creeks and beaver ponds; beaver populations trapped out; cattle and sheep in developed campgrounds and picnic areas; and many livestock

permittees with the attitude that these conditions were satisfactory.

We began working with the livestock permittees to correct these problems. We made gains easily with a few cooperative permittees. However, most of our 34 permittees resisted changes, some fiercely. We continued to make some improvements on all allotments, but Congressional inquiries (formal or informal inquiries of the Forest Service by a Congressman or Senator on behalf of the livestock permittees) and other forms of resistance became a regular occurrence. The situation was allowed to become an explosive issue when we conducted a count of cattle on the Goose Creek Allotment in October of 1989.

We had heard from a reliable source that two of the five permittees had too many cattle on the forest. We conducted an unannounced count at the fall shipping facility, a large set of corrals located on national forest lands. We were accompanied by two State of Idaho brand inspectors and two Forest Service law enforcement officers, who observed the operation. We also had one of my staff members fly the country in an airplane to count cattle that had not been gathered into the facility. The actual count did not show any permittee over the allotted number. However, if the 200 head that were not gathered in were proportioned to all permittees, the two who were suspected would have each been about 30 head over. We never took any action as a result of the count because we were not allowed to expand the numbers for the cattle scattered over the 22 000 ha (55 000 acre) allotment outside the corrals.

Another interesting finding of the count was that the two permittees who were reported to have too many total cattle on the allotment also had a suspiciously high number of calves per cow. The average calf to cow ratio was less than 80% for all permittees. Those two permittees had a 116% calf crop, even after running on the range all summer.

This count so infuriated these permittees and their supporters that an all-out effort was launched to have me transferred from my position as District Ranger. Dozens of events began to unfold, but the most significant was an informal agreement that was made between several Forest Service people, with the livestock industry with other very influential people present, that I would be transferred within 12 months. I was also not supposed to have anything to do with grazing on my own District during that time. I was not aware of this arrangement for several months, but it soon became obvious that something strange was going on. Once I finally confirmed that this agreement had been made, I immediately took action to protect myself. It also became obvious that some higher level Forest Service managers were supporting the agreement.

In February 1990, I filed a whistle blower complaint with the Inspector General's Office of the Department of Agriculture. That filing immediately stopped the attempts to transfer me, and put those within the agency who had made the agreement on the defensive. The claims were thoroughly investigated, and my staff and I were commended in the final investigator's report for performing our duties for the good of the resources and fully within existing direction and laws. No one was found guilty of any wrongdoing in making the agreement to move me, but the resulting publicity made the truth of the matter known to the world and allowed us to move forward with our efforts to improve the resources on the District.

Even before the investigation was conducted, word spread of the agreement. The local media inquired briefly. However, the *High Country News* (May 7, 1990) in Paonia, Colorado investigated and did a five-page story. The investigative reporter called a number of involved Forest Service people, ranchers, Idaho Cattle Association, congressmen and senators, and other local residents. This story was investigated by *The New York Times* (August 19, 1990), *People Magazine* (November 19, 1990), *Insight Magazine* (October 22, 1990), *Outside Magazine* (January 1991), *Audubon Magazine* (March 1991), *Wilderness Magazine* (Spring 1991), *Different Drummer* (Spring 1994), and dozens of other periodicals and newspapers across the country, as well as in Canada and Europe.

The Forest Service received thousands of letters and phone calls urging that we be supported by the agency. We received overwhelming support from most people within the Forest Service, conservation organizations, and the general public. We received very strong support from most of the media. It was obvious to the many media personnel that we were handling important problems for the good of the public and that we were being totally honest with them.

On numerous occasions the media and others requested information from our files under the *Freedom of Information Act*. Several national publications requested copies of the whistleblower complaint and investigation findings before the investigation was even complete. Those requests were filled by the Forest Service as soon as they were completed.

The coverage of these events by the media was key to my being able to continue my work on the Twin Falls Ranger District. It also encouraged public land resource managers across the nation. My own staff were able to move forward with some confidence that our actions to improve livestock grazing practices would be supported.

Of course, a few people from within our agency expressed that I had been a terrible embarrassment to the Forest Service. I was quick to point out that those employees who were willing to compromise the resource and make the agreement to move me, with the very people who were violating the terms of the permits they had signed, were the embarrassment to our agency. We heard from hundreds of people from within and without the agency who were thankful that they had public land managers who were willing to fight for the long-term health of the resources.

Through all these events we learned the value of public support for our programs. We learned the value and power of the media, and that there is nothing to fear from the media if we are honest and obey the law.

The drama has not ended with the passing of the events of 1989 through 1991. A number of attempts have been made by permittees to have me moved or discredited. Some severe problems with permittee compliance with regulations continue. We have taken several penalties on permittees for violations. However, most of our permittees are doing a much better job. Both upland areas and riparian areas have improved, in many areas dramatically. Our working relationship with most of our permittees has also improved significantly. Another part of this success story is that we have not reduced the numbers of livestock for any of our 34 permittees on the 18 allotments. All of the improvements have been gained through insisting on better management practices and the addition of some new facilities.

Problems and Solutions

Why should the public care about livestock grazing on their public lands? Grazing is an appropriate and authorized use on the national forests and other public lands. It helps sustain the traditional western way of life, supports ranching families, helps support local economies, and provides us with meat and other products. But, what levels and types of grazing can be accommodated and still provide for the needs of the other important resources and uses on these lands? What level and types of grazing can the people who own these lands tolerate? How much can we, as public land resource managers, allow and be able to say that we are really managing the public lands for the greatest good for the greatest number of people in the long run? Are we really improving conditions for our future generations, or are we allowing conditions to deteriorate, continually losing options for the future? I don't have the answers to all these questions, but will try to present some of the problems we have faced on the Twin Falls Ranger District, and some of the things we have been able to do to, at least, make some gains.

What are the problems related to public land grazing? From talking with numerous other resource managers and specialists around the west, conditions on the Twin Falls Ranger District are fairly typical of most areas that are grazed by livestock. Cattle and sheep came to our country about 130 years ago. It was thought that the tremendous quantities of forage could never be depleted. At one time, there were 175 000 cattle and 80 000 sheep on what is now the Twin Falls Ranger District and the nearby surrounding areas. They were there continually, except when winter drove them out of the high country. An additional 200 000 sheep were moved through the area every spring and fall, utilizing summer ranges in the mountains north of the Snake River Plain. The District currently permits about 7000 cattle and 8000 sheep. The season of use averages around 4½ to 5 months. The time spent on any particular piece of ground is generally only 20–45 days. Many of these animals spend some additional time on Bureau of Land Management (BLM) lands.

Even with a fraction of the cattle and sheep that once grazed in the District, we still have many livestock-related problems. Resource damage must have been particularly severe after the huge numbers of livestock had been on these lands for a few years in the early days of grazing. Areas near any stream or other water source were severely affected. The removal of vegetation and severe compaction of the soil allowed the water from the yearly intense thunderstorms and spring runoff to rush over the land instead of soaking in. Untold quantities of soil were washed down the streams. Streambeds were downcut, which lowered the water tables. Large, wet meadows dried up and turned from lush willow and sedge wetlands to sagebrush flats.

Yellowstone cutthroat trout (*Oncorhynchus clarki*) and rainbow trout (*Salmo gairdneri*) habitat was destroyed or severely degraded and anadromous fish [chinook salmon (*Oncorhynchus tshawytscha*) and steelhead (*Salmo gairdneri*)] habitat was lost. With the lowering of the water tables and swift runoff, many permanent streams dried up.

Some public land managers and specialists, members of the public, and politicians before us recognized the problems. These people did what they could to reduce livestock numbers and to encourage better management practices. Some people made significant strides toward solving the problems, which can be seen by the reduced numbers now grazing in the Twin Falls Ranger District. However, we are learning more about the needs of all the resources and the wishes of the American public every year. Probably the most important recent changes are related to recognizing the importance of riparian areas. Not many years ago, these were treated as sacrificial areas, while we worked to stabilize and improve the vast uplands. We nonetheless knew that these streams, springs, and ponds had to provide water for livestock, which was essential for utilizing the range resources.

We now recognize, however, that these riparian areas are extremely important and that we can no longer manage them as single-use areas, particularly if the use is degrading other resources and uses. Future grazing will have to take place in such a way that watershed and soil values are protected or improved. Wildlife and fisheries habitat must be maintained and improved. The recreation and aesthetic values of these special places must be improved. In some cases, archaeological sites in and near riparian areas need protection from livestock. These riparian areas comprise only 2–3% of the land area of the west. However, they are key to the survival of most wildlife and fish species, are among the most important recreation areas for people, and provide us with critical domestic and irrigation water. Healthy riparian systems provide larger amounts of better-quality water than can be produced from degraded riparian systems.

I became aware of the importance of riparian areas and the tremendous recovery ability of degraded riparian areas in the spring of 1987. A number of us from the Sawtooth National Forest went to visit Wayne Elmore of the Bureau of Land Management at Prineville, Oregon. His slide presentation and tour of study areas presented dramatic evidence of the resilience of riparian systems. In the 9 years since that time, we have implemented several dozen riparian demonstration projects within the District, many of them with the help of local volunteers, conservation clubs, livestock permittees, the Idaho Department of Fish and Game, Izaak Walton League's Public Lands Restoration Task Force, and volunteers from other federal agencies.

Some of our projects involve significant portions of streams. One riparian pasture contains 6.5 km (4 miles) of an important Yellowstone cutthroat trout (Oncorhynchus clarki) stream. Five others are riparian pastures containing from 2.5 to 5 km (1.5 to 3 miles) of stream. We have fenced a number of areas that now totally exclude grazing to provide examples of the potential of some riparian areas to recover from past degradation. These include two important wildlife pond areas, 2.5 km (1.5 miles) of a large gully with intermittent water, which is 9 m (30 feet) deep in places, and a number of smaller exclosures on creeks and ponds. We have also fenced a number of springs to protect them from trampling. Most of these have been in place long enough that they are showing dramatic improvements. Even 1 year of rest from cattle grazing has resulted in astounding increases in ground cover, stream bank vegetation, and clean water. The ponds are now providing excellent wildlife habitat, with a significant increase in waterfowl production.

Dry Gulch, the 9 m-deep gully I mentioned above, is a classic example of one riparian area that was still unraveling. I have been criticized at times for being too hasty to give cattle the blame for most of our riparian areas degradation. However, we see dramatic improvement in our riparian demonstration areas with the removal or strict control of just one factor–cattle. In some areas, heavy recreation use or other factors are the problem. Almost all of the impacts on the Twin Falls District, however, relate to cattle, and in some cases sheep, where they are occasionally allowed to bed by water. Our timber-cutting activities take place primarily away from perennial streams. We have historical accounts of vast areas that were denuded by livestock in the early 1900s.

Dry Gulch: A Case Study

Dry Gulch is a deep wash, containing intermittent water, that has eroded into ancient deposits of volcanic ash. This soil is so fragile that a piece of this material can melt away into standing water as if it were sugar. As Wayne Elmore's studies have shown, it is nearly impossible to get erosion structures to hold in this fine soil—vegetation is required.

Dry Gulch is about 11 km (7 miles) long, the lower 3 km (2 miles) of which are on BLM and private land. It varies from 3 to 15 m (10–50 feet) in width, and during some high intensity thunderstorms runs water 1.2 m (4 feet) deep. Because of continued removal of vegetation and loosening of the surface soil from hoof action, each storm removes large quantities of material. This has been deposited in downstream private fields, creeks, and one reservoir. Fences that were built in the 1950s

hang in space across vertical banks 6–9 m (20–30 feet) high, which have been cut away in meanders of the creek.

In 1988, one of my Range Conservationists, Ray Neiwert, asked the Soil Conservation Service to measure the physical changes in a portion of Dry Gulch over a period of 30 years. They compared the dimensions of the 1524-m (5000 foot) stretch just above the forest boundary, using our 1957 and 1987 aerial photos. Their measurements showed that we had lost 39 086 m^3 (317 acre feet) of soil from less than 1.6 km (1 mile) of this gully in 30 years. This quantity would roughly equal the removal of 13 dump trucks carrying 8 m^3 (10 cubic yards) of soil every day for 30 years. This was not ancient history; it was still occurring under our present system of management.

My lead Range Conservationist, Ralph Jenkins, directed the fencing of a 2.5-km (1.5 mile) stretch of Dry Gulch in 1988 and 1989, upstream from the forest boundary. This exclosure includes the 1524-m (5000 foot) stretch that was measured as described above. We fenced the cattle out of all of the late-season water, so we provided a trough on the bench to the west by extending an existing pipeline. The plan is to not graze this area at all, as a demonstration of how the gully can be stabilized and possibly rebuilt by catching sediment in the lush vegetation that would proliferate.

Our Forest Biologists, Hydrologist, and Soil Scientist established three permanent monitoring stations within the exclosure, and read them prior to the first year of rest from grazing. They were again read in 1991 after 3 years without grazing. In that short time, this gully had refilled itself by nearly 30 cm (1 foot). The degradation within the exclosure had been reversed, and the rebuilding process had begun. The transects are scheduled for remeasurement this year, but the gully appears to have refilled itself by at least 60 cm (2 feet). The entire gully, except for one steep bedrock section, is filled with a heavy growth of vegetation. More than half the length of this fenced section contains water, and is now filled with lush sedges and grasses, and many thick stands of willow. Before this project, we could find only a dozen clubbed willow plants under 30 cm (1 foot) tall in all of Dry Gulch. Now, they are beyond what anyone can count inside the exclosure. More of the gulch is watered by the small stream than before, because the 60-cm (2 foot) deep sponge of sediment holds water and releases it slowly. The banks of the gulch, except where vertical, are saturated 1.2–2.5 m (4–5 feet) above the water and are now armored with a mat of heavy grasses. This 60-cm (2 foot) increase in the water table has already changed the lower 90 m (300 feet) of a dry side draw into a wet meadow.

When we looked at the BLM's Camp Creek exclosure with Wayne Elmore in 1987, I believe it had refilled itself 2.5 m (8 feet) in 15 years of rest from grazing. It contained a small meandering stream, where a dry flash flood wash had been, and it already contained a good population of speckled dace (*Rhinichthys osculus*). Someday, we expect that Dry Gulch will run a continuous, year-round stream and will support a population of various species of fish. The BLM is also working to improve Dry Gulch below the forest boundary.

Summary

Our goal is to allow grazing on as much suitable range as possible. We cannot eliminate grazing on all riparian areas. It is impossible to fence all the hundreds of miles of stream, and the thousands of springs, ponds, and lakes in the forest. There are also fragile upland areas that are being affected by grazing in several ways. Our challenge is to find the right management strategies for each piece of ground that provides for the many other resources and public needs.

In the Twin Falls Ranger District, we have seen the conditions of most of our riparian areas improve steadily in the past 9 years. Some were improving before that time, as a result of the deferred grazing systems that had been implemented. However, we still have a long way to go before we are totally meeting the established standards and guidelines of our Forest Land and Resource Management Plan, and the needs of the other resources. Some relatively small areas have not improved, and continue to be degraded. We have worked patiently with our livestock permittees to get them to meet their obligations under the terms of their permits. Most of our 34 permittees are doing better. Some are working hard to improve the areas on which they graze and have helped us implement riparian pastures on their allotments. Each year several more permittees have gone from resisting any change to implementing needed improvements. There are only a few individuals who still resist putting adequate effort and resources into management of their allotments to achieve the results we would like to achieve.

We have been able to achieve considerable improvement of our resources even though continued change is still needed. The following are some of the things that have helped us make our gains to date:

- Public pressure to take better care of public lands has resulted in changing policy. Public pressure locally has provided strong support for Forest Service staff to improve grazing practices in this District and the forest. Many people are demanding it.
- National and local media have generally been supportive of our efforts to make the improvements. A considerable amount of controversy has erupted periodically from pressing some permittees to improve their management practices. The resultant publicity has raised the awareness of the American public to the need for changing grazing practices on public lands. I have heard from many people and organizations all of whom express strong support for solving these problems. They were glad to know civil servants were working diligently to manage their resources responsibly.
- Other districts of this forest are tackling some tough range issues, as are many other national forests and state agencies. Our Regional Office staff in Ogden, Utah, and the Washington Office staff are providing support and developing valuable policy to help us meet these challenges. The work of other Forest Service units and agencies supports us in our efforts here.
- The good work that many of our staff began before us has put us in a fairly good position to move on quickly to ecosystem management, and to properly caring for all resources.
- Most of our permittees are improving their management practices. They are doing a better job of moving livestock from unit to unit within allotments when prescribed utilization levels are reached. However, more improvement is needed in this area, so that the permittees make their own moves rather than waiting for us to tell them it is time to move. They are generally cleaning the cattle units thoroughly after they move out, which was not being done on most allotments in the past. Riparian conditions are improving in most areas because they contain cattle for a month or less, instead of all summer and fall. Maintenance of fences and water developments has been much better for the last 6 years. Poor maintenance was contributing to cattle grazing on riparian areas all season. Riparian areas are wet zones (naturally irrigated), and the vegetation that is so important to protecting and improving these areas has a chance to regrow if used early and if cattle are completely removed after use. Permittees are doing a better job of using salt as a tool to attract cattle away from roads and riparian areas. Some permittees are doing a better job of riding in order to distribute cattle out to stock ponds and troughs and away from riparian areas. Lack of proper herding, however, is still one of our most serious problems on most allotments. This lack of herding not only impacts riparian areas but also results in over-used areas, while many usable acres farther from water remain practically untouched. This forces us to move the herd through the area faster than would be necessary with good herding in order to try to meet our utilization guidelines in the Forest Plan.
- We have installed additional water developments to provide more water for cattle away from riparian areas. Most of these are ponds with the spring source fenced.
- The riparian demonstration projects we have established are proving to be invaluable. We now have many examples that show the potential of riparian areas and uplands to recover. We finally have some good answers when we are told that a certain severely affected stream is better than it has ever been before. These demonstration areas have greatly increased the understanding of riparian systems for our permittees, other agency staff, the public, and our own employees. All of the Districts of the Sawtooth National Forest now have good examples in place, as does the BLM.
- Many of our riparian demonstration projects have been constructed with the help of partners who share a commitment to what they demonstrate. Some of these facilities are being maintained by local conservation organizations. Dozens of tours of these areas have been conducted by District staff and others who helped establish them. We also have a good before-and-after photo record of most of these project areas.
- We have reintroduced beavers (*Castor canadensis*) into many areas where they no longer existed. Existing beaver populations have also increased as a result of the greatly improved conditions. Willow, aspen, cottonwood, and sedges have increased with improved management so that viable beaver populations can thrive. In our South Fork Shoshone Creek riparian pasture, regenerating willow plants counted on three transects increased from 30 to over 400 in 3 years during the initial years of rest from cattle grazing. The beaver ponds have increased from none

to about 30 in this area, and the willow continues to increase faster than the beavers can use it. The raised water table has also increased the size of the riparian area. We have completed 2 years of controlled early spring grazing in this riparian pasture. The riparian and upland conditions continue to improve during both the use years and the rest years.

The beavers have raised the water table in a number of areas as much as 2 m (6 feet). We have had to build up sections of road with gravel where beaver activity has flooded or saturated roads. We believe the expanded wetlands are important enough that we will not remove dams if there are other alternatives.

The Idaho Fish and Game Commission has also helped us improve our riparian areas by closing about one-third of our streams to beaver trapping. This closure began at our request in 1989 and has been extended every 2 years since then. I recently asked that three more major drainages be closed to trapping to protect important beaver populations in the District.

It is often said that beavers will eat themselves out of food and building materials and will have to move on, leaving the dams to deteriorate and wash out. Some of our livestock permittees say they have seen this happen many times. We believe this was rarely the case under natural conditions. Improved grazing practices in our District have resulted in prolific sprouting of willow, aspen, and cottonwood in almost all of our streams, even outside our demonstration areas and riparian pastures.

Beavers feed in the spring and summer almost entirely on sedges and forbs if available. The improved riparian conditions make those plants more available to the beavers, removing the necessity for them to deplete the woody species all year long.

- Our local public has become very aware of the needed changes concerning range management of the forest. Some individuals have been diligent in reporting and photographing affected areas. They have also applied significant pressure on the agencies and politicians to encourage us to do a better job. Some have taken advantage of the *Freedom of Information Act*, making numerous requests for copies of correspondence and range inspection notes from allotment files. Some have requested and been allowed to attend the spring annual operating-plan meetings with the permittees, for the past 6 years. This created considerable concern on the part of some permittees in the first year. A few permittees still object to allowing the public to attend these meetings, although some of their own associates have always attended. We have had no disruption of the meetings by the public, and have been able to conduct all needed business with the permittees. Some media coverage has occurred, but has generally been informative and positive. The public has been made much more aware of the allotment management process, and we have been able to demonstrate openness with the public as we manage their lands.

  The improvements of both riparian areas and uplands is dramatic and widespread enough that many forest users have commented favorably on the changes.
- We have worked closely with Congressional Aides and our County Commissioners to gain an understanding of our Forest Plan direction and the needs of the resources.
- We have worked continually with our permittees to make the needed resource improvements, while trying to minimize the impacts upon them or the local communities. Many tours of the allotments have helped us, the permittees, and the public to gain a better understanding of each other, and the needs of the resources. Many of the permittees now can recognize and are proud of the improvements they have made to the resources. Some have commented on the increased late-season water flows they now see below streams with numerous beaver ponds.

Many of our public land managers are working hard to improve livestock grazing practices that affect our riparian systems and uplands. Others, however, seem to feel they must wait until they have the ultimate in back-up data and support. Most of us are working under guidelines that allow us to make some significant gains now, if we are willing to take some risks for the environment. Improvements need to take place faster than they are occurring in many areas. The situation is such that we must make significant improvements now before we lose more of our basic building-block resources of soil and water.

These lands belong to all the owners of the public lands, every individual in the nation. We must ask ourselves as public land managers whether we are making significant improvements for future generations. Or, are we allowing their resources to continue

to deteriorate, eliminating options for the future? These very questions are behind our national policies, which seek to improve range land and riparian area conditions, and to practice true ecosystem management. Livestock permittees are also important partners in the management of our rangelands, and their needs should be accommodated as much as possible. Most of them realize that their livelihoods depend upon the future health of the rangelands and riparian areas. More individuals are now aware of what is needed to improve the resources on their public land allotments, and most are working toward that end.

I recently heard one of our Forest Supervisors say that his management philosophy can be summed up as: "Just keep the soil from going down the creek". I think that about says it all. The encouraging thing for me is that with proper management we can actually see an amazing rebuilding process take place in our riparian areas. This is not only possible and necessary, but also rewarding! What a privilege we, as public land managers, have been given by the American people to manage these beautiful areas for them! Let us do a job of which they and we can be proud.

After the 1992 riots in Los Angeles, the people who beat up the truck driver (which we saw on our television sets) were arrested with the help of a witness who came forward and turned them in. When she was asked in a television interview why she took that risk, she said, "It is within everyone's power to do nothing". I think that quote also applies to the management of our public lands.

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Biology and Mercury Content of Bull Trout in the Oldman River Reservoir and Adjoining Waters

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Abstract

This report summarizes findings from a 5-year project to monitor mercury concentrations and conduct supplemental inventories on fish populations in the Oldman River Reservoir following its creation in 1991. Reservoir fish populations were sampled in the fall of each year (1991–1995) using standardized gill-net procedures. Overall fish abundance in the reservoir was high throughout the period of study; yearly catch-per-unit-effort for all species combined ranged from 3.5 to 10.7 fish·h^{-1}·100 m^{-1} net. Catch rates for bull trout ranged from 0.27 to 0.94 fish·h^{-1}·100 m^{-1} net, suggesting that significant numbers of this species continue to exist upstream of the dam. The reservoir appears to harbor a suitable food base for bull trout in the form of populations of longnose sucker, white sucker, and mountain whitefish. Prey items found in the stomachs of 80 bull trout collected from the reservoir consisted primarily of fish, hence bull trout appear to be using the reservoir food base. It is hoped that bull trout will adjust to impoundment of the upper Oldman River system by adopting an adfluvial life history pattern. Although muscle mercury concentrations in bull trout from the Oldman River Reservoir increased significantly from 1991 to 1995, mean total mercury concentrations remained below the 0.5 mg·kg^{-1} guideline for safe consumption recommended by Health Canada. Other common fish species (rainbow trout, mountain whitefish, longnose sucker, and white sucker) sampled during this study from the Oldman River Reservoir, and from upstream and downstream sites, also had mean mercury concentrations below 0.5 mg·kg^{-1}. Continued monitoring of fish populations in the Oldman River Reservoir is recommended because limnological and biological conditions within newly formed reservoirs commonly require longer than 5 years to stabilize.

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Introduction

Southern Alberta's Oldman River Reservoir was created in 1991 by construction of an earth and rock filled dam near the confluence of the Oldman, Crowsnest, and Castle rivers. The reservoir is expected to affect local fish populations, notably populations of bull trout (*Salvelinus confluentus*), a species that is currently considered of special concern in Alberta (Berry 1994). There is also concern that the reservoir may lead to accumulations of mercury in fish. Flooding of soil generally enhances the microbial conversion of ubiquitous inorganic mercury to methyl mercury, which is highly toxic and readily accumulates in aquatic food chains (Jackson 1988). Elevated mercury concentrations in fish have been reported in several reservoirs in Canada (Bodaly et al. 1984; Jackson 1988), and elsewhere (Potter et al. 1975; Lodenius et al. 1983). To address these concerns, the Alberta Government implemented an extensive environmental monitoring and fisheries mitigation program for the reservoir. As part of this program, the Alberta Research Council[1] (ARC) has, since 1991, monitored mercury concentrations in fish from the Oldman River Reservoir and from adjoining waters. Although the primary purpose of this program is to monitor mercury concentrations, the data gathered also provide useful supplementary information on the biology of reservoir fish populations.

This paper presents a summary of the information gathered during the study period (1991–95) on bull trout in the Oldman River system. The information presented on bull trout includes mercury concentrations in muscle, catch-per-unit-effort, age and size distribution, and diet composition.

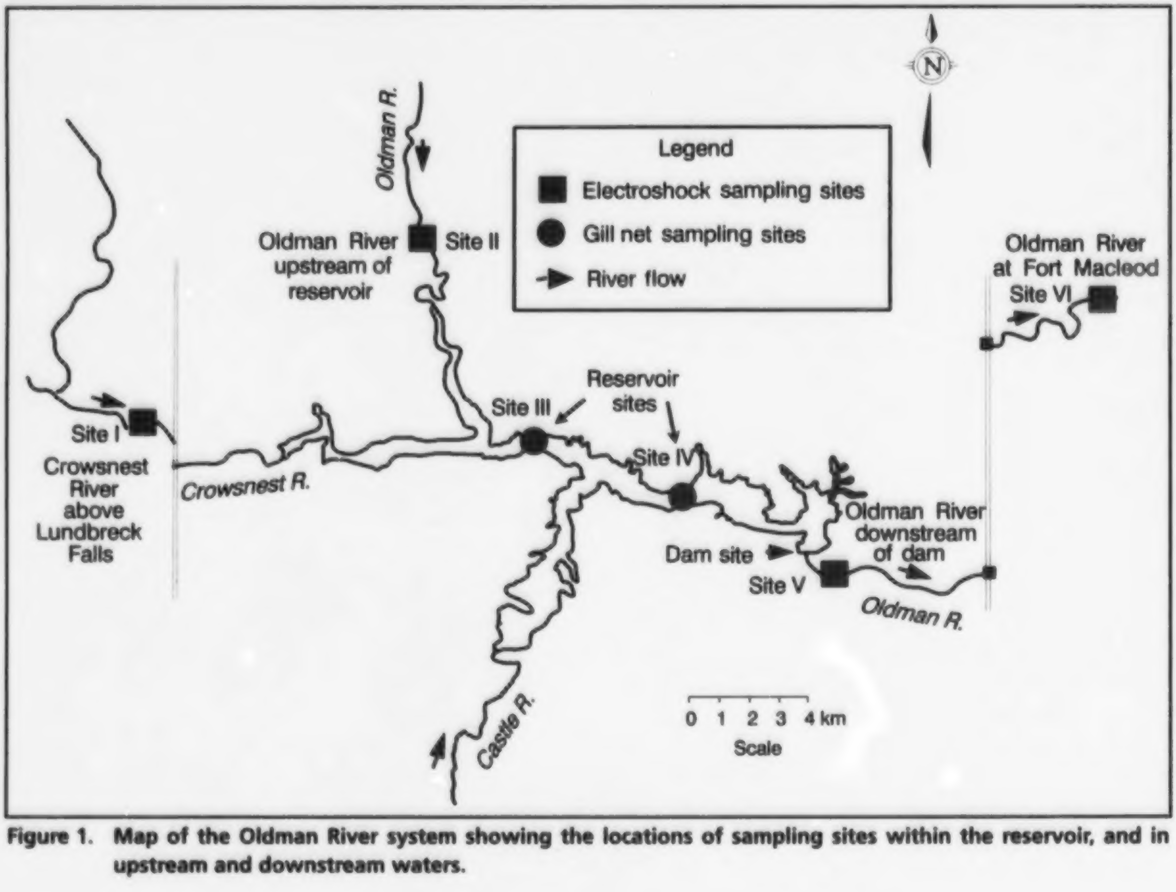

Figure 1. Map of the Oldman River system showing the locations of sampling sites within the reservoir, and in upstream and downstream waters.

[1] The work reported here was conducted by the Alberta Environmental Centre, which joined with the Alberta Research Council on July 10, 1996.

Methods

Fish populations within the reservoir, and in upstream and downstream waters, were sampled in late September or early October of each year from 1991 to 1995, inclusive. The collection sites upstream of the reservoir were the Crowsnest River (about 15 km above Lundbreck Falls), and the Oldman River near the Olin Creek bridge (Fig. 1). Sampling within the reservoir was conducted at one site west and one site east of the centrally located Castle River arm. Downstream of the reservoir, fish were collected from the Oldman River immediately below the dam, and from a site near Fort Macleod, approximately 50 km east of the reservoir.

The reservoir sites were fished using 100 m (1.8 m multipanel monofilament gill nets consisting of nine different mesh sizes ranging from 25 to 127 mm. The locations and methods of gill-net sampling were held as constant as possible from year to year to ensure data continuity (Moore et al. 1993). Standardized gill-net sampling consisted of fishing with two nets simultaneously; one being set in shallow water (2–10 m), and the other in deeper water (10–30 m). Both nets were suspended immediately above the bottom and set perpendicular to the shoreline. In 1991, gill nets were set overnight to make the sampling consistent with a previous study conducted at the Dickson Dam Reservoir (Moore 1989a, b). However, this practice was stopped in 1992 after catch rates at the Oldman River Reservoir proved to be higher than expected. Beginning in 1992, gill nets were set for only 3 to 8 hours to avoid catching excessive numbers of fish, particularly bull trout.

After the standardized gill-net sampling had been completed, it was usually necessary to conduct additional sampling (1992 to 1995 inclusive) to fill in gaps in desired sample sizes for certain species. This supplemental gill-net sampling was conducted using only the five largest mesh sizes (76–127 mm), and only in shallow water. Sampling of fish from sites upstream and downstream of the reservoir was done by electroshocking from boats. Fish were placed in a freezer within a few hours after capture, kept at –20°C, and transported frozen to Alberta Research Council (ARC), Vegreville for dissection and processing.

Data collected from each fish included fork length and total body weight, sex, diet composition as determined from stomach contents, and age. Bull trout were aged from otoliths as recommended by Mackay et al. (1990). The ageing was done by Mr. W. English (Alberta Environmental Protection, Fisheries Management Division) who is experienced in interpreting fish ageing structures and familiar with the biology of fish populations in the Oldman River system. A detailed validation of bull trout ages by methods such as collection of 0- and 1-year-old fish to verify the position of the first annulus, and seasonal sampling to verify the time of annulus formation, was not conducted owing to stock conservation concerns. Nevertheless, it was considered worthwhile to present data on bull trout ages here as an interim measure to aid in the design of more definitive studies.

Samples of epaxial white muscle were excised from a selected subsample of fish for measurement of total mercury concentration. Total mercury concentration was measured by digesting muscle samples in concentrated sulphuric and nitric acids to convert organic and inorganic mercury to divalent mercury ions, reducing the divalent mercury ions to elemental mercury with stannous chloride solution, and measuring the resulting mercury vapor with a cold-vapor mercury analyzer (Moore et al. 1993).

To evaluate the significance of year-to-year changes in mercury concentrations it was necessary to combine data from some sampling sites for the purpose of achieving adequate sample sizes. The locations at which year-to-year comparisons of mercury concentrations were made are upstream of the reservoir, the reservoir itself, and downstream of the reservoir. Because sample sizes were reduced after 1991, the mean fork length of bull trout used for mercury analysis varied somewhat among years. Therefore, it was necessary to test the significance of regression of mercury concentration on fork length to determine whether length was needed as a covariate in the model used to test for year-to-year variations in mercury concentrations. If significant regressions between mercury concentration and length occurred in any year, tests for the equality of regression coefficients were performed to determine whether a common slope or an unequal slope analysis of covariance (ANCOVA) model was more appropriate for testing year-to-year variations in mercury concentrations. When an unequal slope ANCOVA was used, comparison of mercury concentrations among years was done at the overall mean length of bull trout. The above analyses were performed separately for each location using the MIXED procedure of the SAS® statistical program as described in Littell et al. (1996).

Table 1. Fish species composition in the Oldman River Reservoir as indicated by standardized gill-net catches. Data from both reservoir sampling sites and both sampling depths are combined.

Species	Percentage by number of total catch 1991 (n = 600)	1992 (n = 98)	1993 (n = 64)	1994 (n = 155)	1995 (n = 129)
Bull trout	7.3	7.1	23.4	5.2	4.7
Rainbow trout	3.0	1.0	1.6	0.0	2.3
Brown trout	0.5	0.0	0.0	0.7	0.8
Mountain whitefish	24.8	43.9	48.4	21.3	11.6
Longnose sucker	57.8	41.8	20.3	60.0	59.7
White sucker	6.5	6.1	6.3	12.9	20.9
Total (all species)	100	100	100	100	100

Table 2. Catch-per-unit-effort for bull trout collected from the Oldman River Reservoir using standardized gill-net procedures

Year	Total number of bull trout caught	Number of bull trout caught per hour per 100 m of net: West site	East site	Both sites combined
1991	44	0.63	0.24	0.43
1992	7	0.27	0.26	0.27
1993	15	0.69	1.19	0.94
1994	8	0.43	0.65	0.54
1995	6	0.29	0.27	0.28
Mean		0.46	0.52	0.49

Table 3. Size-at-age of bull trout from the Oldman River Reservoir and adjoining waters. Fork length and body weight are expressed as mean ± SEM.

Site	Parameter	Age 2	3	4	5	6	7
Reservoir	Number of fish	17	31	16	8	1	4
	% of total	22.1	40.3	20.8	10.4	1.3	5.2
	Fork length (cm)	26.6 ± 0.5	29.8 ± 0.6	30.0 ± 1.9	34.8 ± 1.0	34.6	47.3 ± 5.0
	Weight (g)	206 ± 12	287±16	339 ± 94	405 ± 34	401	1136 ± 300
Upstream of reservoir	Number of fish	1	0	3	1	4	0
	Fork length (cm)	24.2	–	37.0 ± 6.8	36.0	42.6 ± 1.7	–
	Weight (g)	142	–	453 ± 180	450	897 ± 143	–
Downstream of reservoir	Number of fish	0	5	4	3	1	0
	Fork length (cm)	–	34.2 ± 2.2	41.4 ± 3.1	38.9 ± 4.5	47.0	–
	Weight (g)	–	453 ± 82	791 ± 189	710 ± 263	1040	–

Table 4. Diet composition of bull trout from the Oldman River Reservoir

Item	1991 (43 stomachs examined) Number of stomachs	Percentage occurrence	1992–1995 (37 stomachs examined) Number of stomachs	Percentage occurrence
Empty	17	39.5	25	67.6
Fish	17	39.5	12	32.4
Cladocera	9	20.9	1	2.7
Insecta	4	9.3	0	0.0
Oligochaeta	1	2.3	0	0.0
Nematomorpha	1	2.3	0	0.0
Unidentified matter	1	2.3	0	0.0

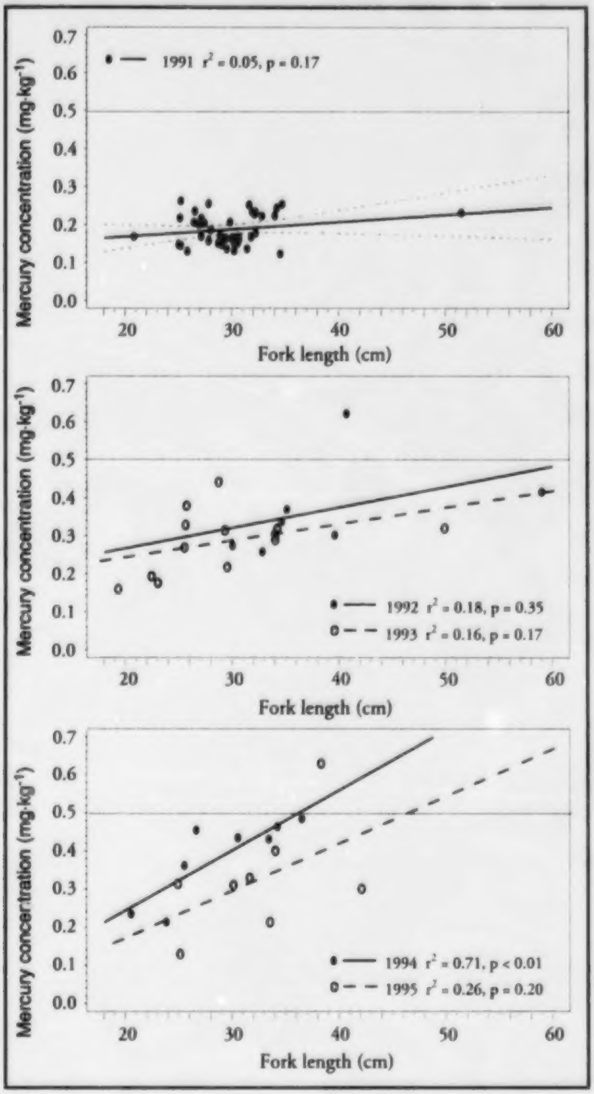

Figure 2. Scatter plots showing the relationship between muscle mercury concentration and fork length in bull trout from the Oldman River Reservoir. Regression lines are fitted to the data separately for each year and 95% confidence limits are shown for 1991. Confidence limits for other years are omitted for clarity. Coefficients of determination (r^2) and probability values for the significance of linear regression are shown for each year. The 0.5 $mg{\cdot}kg^{-1}$ safe consumption guideline is also indicated in each panel.

Results

Species Composition in the Reservoir

Longnose sucker (*Catostomus catostomus*) and mountain whitefish (*Prosopium williamsoni*) were the predominant species represented in the standardized gill-net catches. The proportion of mountain whitefish in the catch declined between 1993 and 1995, whereas the proportion of white sucker (*Catostomus commersoni*) increased (Table 1). Bull trout were the most frequently caught trout species, comprising 23.4% of the catch in 1993, and 4.7 to 7.3% in other years. Three lake trout (*Salvelinus namaycush*) were captured during non-standardized gill-net sampling in the reservoir, two in 1992, and one in 1995 (data not shown). Very few rainbow trout (*Oncorhynchus mykiss*) and brown trout (*Salmo trutta*) were caught from the reservoir. Rainbow trout appear to have been under represented in the gill-net catch when compared to the frequency with which anglers were catching this species at the reservoir (Ripley 1995). Reasons for this discrepancy are not known for sure, but perhaps rainbow trout are more abundant in the immediate vicinity of the river mouths, where we did not set nets, or maybe they prefer the upper water column rather the bottom where our nets were set.

Catch-per-unit-effort

Average yearly catch-per-unit-effort (CPUE) varied from 0.27 to 0.94 bull trout $\cdot h^{-1} \cdot 100\ m^{-1}$ net (Table 2). There were no consistent differences in bull trout catch rates between the two sampling sites in the reservoir, or between the shallow and deep net locations (data not shown). The CPUE data are not sufficiently extensive to allow reliable conclusions regarding year-to-year variability in bull trout abundance within the reservoir.

Age and Size Distribution

Eighty-three percent of the bull trout captured from the reservoir were from 2 to 4 years of age, and the remainder were ages 5 to 7 (Table 3). Seven-year-old bull trout collected from the reservoir had attained an average body weight of more than 1 kg. Bull trout collected by electrofishing upstream and downstream of the reservoir were larger, at any given age, than those collected from within the reservoir by gill-net sampling.

Table 5. Muscle mercury concentrations of bull trout from the Oldman River Reservoir system. Data are expressed as mean ± SEM. Fork length is in centimeters and total mercury concentration in mg·kg⁻¹.

Parameter	Year 1991	1992	1993	1994	1995	Statistical procedure	P value[a]
			Upstream of reservoir on Oldman River				
n	3	4	2	1	0		
Fork length	38.2 ± 7.0	34.9 ± 3.9	39.8± 1.6	46.0	–		
[THG] arithmetic mean	0.230 ± 0.036	0.272 ± 0.062	0.366 ± 0.131	0.400	–		
[THG] LS mean[b]	0.228 ± 0.063	0.297 ± 0.056	0.352 ± 0.077	0.339 ± 0.116	–	Common slope ANCOVA	0.64
			Oldman Reservoir				
n	43	7	13	8	8		
Fork length	29.9 ± 0.7	38.8 ± 3.6	29.3 ± 2.2	28.9 ± 2.0	32.5 ± 2.1		
[THG] arithmetic mean	0.187 ± 0.006	0.369 ± 0.047	0.286 ± 0.023	0.385 ± 0.037	0.329 ± 0.052		
[THG] LS mean[c]	0.189 ± 0.011[A]	0.325 ± 0.036[BC]	0.292 ± 0.020[B]	0.414 ± 0.027[C]	0.307 ± 0.026[B]	Unequal slope ANCOVA at overall mean length (30.7 cm)	<0.01
			Downstream of reservoir on Oldman River				
n	9	5	0	0	0		
Fork length	34.0 ± 1.5	45.1 ± 1.2	–	–	–		
[THG] arithmetic mean	0.249 ± 0.020	0.339 ± 0.020	–	–	–		
[THG] LS mean[b]	0.248 ± 0.025	0.342 ± 0.040	–	–	–	Common slope ANCOVA	0.13

[a] Indicates the significance of between-year variability in total mercury concentration as tested using the indicated statistical procedure.

[b] Represents the least squares mean of total mercury concentration as computed in conjunction with ANCOVA analysis. The least squares means are estimates of what mean total mercury concentrations would be if the fork length of bull trout (i.e., the covariate) had been held constant at its mean value.

[c] Superscripted letters indicate the results of pair-wise comparisons of least squares mean mercury concentrations (Littell et al. 1996). Least squares means designated by the same letter are not significantly different at $p = 0.05$.

Diet Composition

The diet of bull trout in the reservoir consisted primarily of fish (Table 4). Of the 56 fish found in bull trout stomachs, only 10 were sufficiently undigested to allow positive identification. Eight of these fish were salmonids (trout or whitefish), one was a sucker, and one was a burbot. Invertebrates may have been slightly more important in the diet of bull trout in 1991 than in later years.

Total Mercury Concentrations in Bull Trout Muscle

Up to and including the most recent sampling date of September 1995, mean total mercury concentrations in bull trout from the Oldman River Reservoir (Table 5) remained below 0.5 $mg \cdot kg^{-1}$, which is the guideline for safe human consumption recommended by Health Canada. However, statistically significant increases in muscle mercury concentrations of bull trout have occurred within the reservoir since 1991. Mercury concentrations in 1991 were significantly lower than concentrations in all other years. Total mercury concentrations did not differ significantly between 1992, 1993, and 1995, nor between 1992 and 1994.

Mean mercury concentrations in bull trout collected upstream and downstream of the reservoir were also below the 0.5 $mg \cdot kg^{-1}$ consumption guideline. Sample sizes at collection sites upstream of the reservoir were small, and bull trout captured downstream of the dam after 1992 were released alive due to stock conservation concerns. The limited sample sizes at these locations made it impossible to conduct proper regression analyses to determine whether ANOVA, common slope ANCOVA, or unequal slope ANCOVA provided the best model with which to test for year-to-year changes in mercury concentrations. It was decided to use common slope ANCOVA to make between-year comparisons of mercury concentrations at these locations in the belief that some adjustment for variations in fish size was necessary, especially downstream of the reservoir. There is an indication that mercury concentrations in bull trout above and below the reservoir may have increased after 1991, but these changes are not statistically significant (Table 5).

Figure 2 shows the relationship between mercury concentration and fork length in bull trout from the reservoir. The increase in mercury concentration that occurred following impoundment is again apparent. Additionally, the slope of the mercury concentration versus fork length relationship appears to increase after 1991, although the regression is significant only in 1994, when it explained 71% of the total variation in mercury concentrations.

During this study, total mercury concentrations were also measured in several other fish species common in the reservoir and in adjoining upstream and downstream areas. At all sampling sites and years, mean mercury concentrations in these species (rainbow trout, mountain whitefish, longnose sucker, and white sucker) were also below the 0.5 $mg \cdot kg^{-1}$ guideline (Wu et al. 1996).

Discussion

Although a zero bag limit for bull trout was implemented in Alberta in 1995 (i.e., catch-and-release fishing only), bull trout are considered important indicators of mercury contamination because they are the most piscivorous of the common fish species in the reservoir. Piscivorous fish tend to harbor higher concentrations of mercury than those feeding on invertebrates (Potter et al. 1975; Lodenius et al. 1983; Bodaly et al. 1984), and this pattern seems to hold true in the Oldman River Reservoir. Bull trout in the reservoir generally had higher mercury concentrations than invertebrate or bottom feeding species such as mountain whitefish, longnose sucker, and white sucker (Wu et al. 1996).

In the first 5 years after filling of the Oldman River Reservoir, mercury concentrations in bull trout (and other reservoir fish species) had not risen to the levels observed at some other reservoirs such as those in the Churchill River diversion in Manitoba (Bodaly et al. 1984). Factors that may have contributed to the relatively low mercury concentrations in fish from the Oldman River Reservoir include the removal of riparian vegetation and top soil prior to reservoir filling (Wu et al. 1996); the alkaline pH of reservoir water, which favors the volatilization of organic mercury (Stein et al. 1996); and the relative paucity of piscivorous fish species. In reservoirs that are known for mercury accumulations in fish, the highest concentrations occur in highly piscivorous species such as walleye and northern pike (Lodenius et al. 1983; Bodaly et al. 1984). Neither of these species is present in the Oldman River Reservoir. Even bull trout, the only highly piscivorous fish species in the reservoir, spends its early years in tributary streams (Fraley and Shepard 1989), and only a portion of its adult life in the reservoir. Consequently, the potential for mercury accumulation in bull trout would not be as great as in piscivorous fish that live their entire lives within a reservoir.

There is some uncertainty about whether increases in mercury concentrations in bull trout after 1991 can be definitely attributed to the reservoir. Because bull trout generally spend from 1 to 4 years in nursery

streams before migrating downstream to their adult habitat (Fraley and Shepard 1989), it is possible that the smaller bull trout we collected from the reservoir may have spent only a very brief time there before being caught. If this were true, then increased mercury concentrations in these fish after 1991 would have to be attributed to changes in the upstream environment rather than to construction of the reservoir. Nevertheless, construction of the reservoir was the obvious change that occurred in the upper Oldman River system in recent years; and it remains the most likely source for the elevated mercury concentrations observed in bull trout after 1991.

The relatively high catch rates reported for bull trout in this study (Table 5) suggest that significant numbers of this species continue to exist upstream of the Oldman River Dam. However, detailed inferences about bull trout abundance from CPUE data are not warranted due to several confounding issues. One important issue is whether bull trout, being piscivorous, are attracted to gill nets by the presence of entangled fish. If this occurred in our study, then our catch rates would over-estimate bull trout abundance. On the other hand, because we sampled the reservoir in fall, when mature bull trout are spawning in the upper tributaries, our catch rates from the reservoir may underestimate the species' abundance. This consideration is particularly applicable to age classes older than 5 years; the latter being the youngest age at which bull trout commonly become sexually mature (Fraley and Shepard 1989; Nelson and Paetz 1992).

Although our CPUE values for bull trout seem relatively high, these values were not influenced by a sport-fish transport program that was conducted as part of the fisheries mitigation plan for the Oldman River Dam. The purpose of the fish transport program was to collect sport fish trapped below the newly constructed dam and release them several kilometres above the dam where they would have access to upstream spawning areas (Environmental Management Associates 1992, 1994). The fish transport program was implemented during the open water periods of 1989 to 1992, inclusive. In total, 161 bull trout from below the dam were transferred into the reservoir during the 4-year transport program, and all of these bull trout were marked with Floy® tags (Environmental Management Associates 1992, 1994). If our gill-net catch from the reservoir consisted predominantly of transported bull trout, it could mean that few adult bull trout were present above the dam when it was being built. However, of the 80 bull trout collected from the Oldman River Reservoir during our 5-year study (Table 2) only one fish (captured in 1991) bore a Floy® tag. Therefore, our catch rates of bull trout from the reservoir are indicative of populations that existed upstream of the dam when the dam was being built, plus contributions through reproduction and recruitment. The continued presence of small bull trout (<25 cm in fork length) in our catch from above the dam up to and including 1995 (data not shown) suggests that at least some recruitment has occurred after construction of the dam.

It is not known why CPUE for bull trout was particularly high in 1993 (Table 2), or why mercury concentrations were relatively low that year (Table 5). However, the absence of tagged bull trout in our catch from the reservoir in 1993 rules out the sport-fish transport program as an explanation for these observations.

Bull trout are difficult to age reliably. Some authors recommend using otoliths (Mackay et al. 1990), whereas others recommend scales (Goetz 1989). Agreement in age between scales and otoliths appears to be good up to age 3, but poor in older fish (Fraley and Shepard 1989; Goetz 1989). We believe size-at-age data of bull trout collected from the reservoir by gill-net sampling offers the best basis for comparison with published growth rates. Size-at-age of bull trout collected above and below the reservoir should not be used to infer mean growth rates because these fish were captured by electroshocking, which is known to select for larger fish (Reynolds 1996). Alberta bull trout populations for which published growth rates are available include those in Pinto Lake (Carl et al. 1989), the Muskeg River (Boag 1987), Clearwater River (Allan 1980), and the Bow River (Miller 1949). If the ages reported in the present study are correct, the growth rate of bull trout in the Oldman River Reservoir could be greater, up to age 4, than rates reported for other Alberta populations (cf. Carl et al. 1989).

Several factors could explain the apparent fast growth of our fish, including failure to validate age (in our study and in others). Underestimation of age by 1 year would bring growth rates of bull trout in the Oldman River Reservoir more in line with previous studies. Alternatively, fast growth could be a reflection of size dependent migrational behavior. If faster growing juvenile bull trout move to the reservoir earlier than slower growing fish, the apparent growth rate in the reservoir would be biased upwards. Validation of bull trout age and growth rate in the Oldman River system appears to be a useful area for future research and should include

collection of 0- and 1-year-old fish to verify the location of the first annulus.

To some extent, expectations regarding fish abundances in the Oldman River Reservoir were based on observations at other Alberta foothill reservoirs such as the Dickson Dam Reservoir near the city of Red Deer. From 1983 to 1987, the Alberta Environmental Centre conducted fish inventories at the Dickson Dam Reservoir using similar types of gill nets and gill-net procedures, except that nets were set for 24 hours as opposed to 3 to 8 hours in the present study (Moore 1989a, b). Even allowing for lower gill-net catch rates during the night, overall fish abundance in the Dickson Dam Reservoir has been exceedingly low, as indicated by yearly CPUE averages for all species of 0.12 to 0.47 fish$\cdot$h$^{-1}\cdot$100 m^{-1} net (Moore et al. 1989b). In contrast, overall fish abundance in the Oldman River Reservoir has been much higher; yearly CPUE averages for all species combined ranged from 3.5 to 10.7 fish$\cdot$h$^{-1}\cdot$100 m^{-1} net (data not shown). High catch rates from the Oldman River Reservoir were observed right from the start of reservoir operation in 1991. This indicates that the relatively high abundance of fish in the Oldman River Reservoir is largely attributable to pre-existing fish populations rather than to a "trophic upsurge" phenomenon, which is commonly believed to characterize reservoir productivity during the first several years of operation (O'Brien 1990). The high productivity of the Crowsnest, Oldman, and Castle rivers, and nutrients contributed by communities in the Crowsnest Pass, may also have helped maintain abundant fish populations in the Oldman River Reservoir in recent years.

In summary, populations of bull trout and other gamefish in the Oldman River Reservoir appear to have remained relatively abundant during the first 5 years of reservoir operation. Mean concentrations of mercury in reservoir fish populations have not exceeded the federal guidelines for consumable fish flesh. However, because limnological and biological conditions within newly formed reservoirs commonly require longer than 5 years to stabilize, continued monitoring of fish populations in the Oldman River Reservoir is recommended.

Acknowledgments

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The Contribution of Forest Litterfall to Phosphorus Inputs into Lake 239, Experimental Lakes Area, Northwestern Ontario

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Abstract

In conjunction with a catchment scale study of the effectiveness of current Ontario buffer strip guidelines on fish habitat, phosphorus (P), nitrogen (N), and carbon (C) inputs to lakes via litterfall from the forest edge are being measured at the Experimental Lakes Area (ELA), in northwestern Ontario. Phosphorus inputs from 100-year-old and fire regeneration jack pine stands along the shores of Lake 239, a headwater lake at the ELA, are the subject of this paper. In the summer of 1995 pans containing distilled water were set out in two pairs of transects from the shore to the center of the lake. Samples were collected at 2 to 4 day intervals from mid-May to mid-October. After the identification of large particles, the samples were analyzed for total P, N, and organic C. Vegetative material, spiders, insects, and fecal material were collected in the pans. Jack pine pollen dispersal in late June to early July was responsible for the largest input; it was also the most broadly distributed input over the whole lake. Insects comprised a major part of the litter, were found in samples throughout the summer, and were widely distributed across the lake. Some had presumably emerged from the lake and thus were not a gross addition to the lake nutrient load. Numerically, however, most insects collected were of terrestrial or stream origin. Phosphorus analysis of a small subset of litter pans sampled over the open water season on Lake 239 has been completed. Phosphorus input to the lake by litterfall was 1.2 $\mu M \cdot m^{-2} \cdot d^{-1}$, about 50% more than P measured in precipitation during the litterfall sampling periods. The rate of P deposition by litterfall within 9 m of shore was six times the mean rate over the rest of the lake surface. Over half the total P deposition by litterfall on the lake occurred within 30 m of shore.

McCullough, G.K. 1998. The contribution of forest litterfall to phosphorus inputs into Lake 239, Experimental Lakes Area, northwestern Ontario. Pages 159–168 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

Phosphorus (P) is a major limiting nutrient in many northern latitude lakes. Hecky et al. (1993) showed that P deficiency is the most important limitation to productivity in small oligotrophic Canadian Shield lakes. Investigations of P input to lakes have generally concentrated on two pathways: stream and overland runoff, and precipitation (Schindler et al. 1976; Campbell 1994). Where P retention (as calculated by mass balances) has been low compared to P sedimentation rates, unmeasured aeolian inputs have been suggested as a cause (Evans and Rigler 1980; Cole et al. 1990).

From the 20 year balance for Lake 239 (Bayley et al. 1992) at the Experimental Lakes Area (ELA), P sedimentation is 1500 $\mu M \cdot m^{-2} \cdot y^{-1}$. The P sedimentation rate derived from a center basin core was 8400 $\mu M \cdot m^{-2} \cdot y^{-1}$, calculated from data reported by Kipphut (1978), assuming no long-term sedimentation occurred above the 6 m isobath (i.e., a focusing factor of 1.5), or six times the balance-determined rate. Nor was this discrepancy unusual. Figure 1 shows corresponding ratios derived from published data plotted against total measured P input for several Shield lakes in southeastern Ontario. With several exceptions, lakes with larger unit area P inputs tended to have better determined balances, as judged against the P sedimentation rate determined by coring.

Several studies (Rau 1976; Kowalczewski and Ryback 1981; Cole et al. 1990) have shown litterfall from the forest edge to be an important contributor of P to small headwater lakes. Measured runoff P, a major source of P in lakes, is partly a function of the area contributing runoff directly to the lake. Watershed export is generally reported on a per unit drainage area basis; loading models typically express loading in part as a function of drainage area (Dillon and Kirchner 1975; Bayley et al. 1992). Litterfall from the forest edge (local aeolian input) expressed per unit area of lake surface could be expected to be dependent on the ratio of shoreline length to lake area, but independent of contributing runoff area. Litterfall would increase in magnitude as a source of P per unit lake area with decreasing lake size, and increase in importance relative to runoff as a source with decreasing runoff area. This line of reasoning spurred interest in the measurement of litterfall inputs to ELA lakes.

In 1993, the Ontario government began a watershed cutting experiment to determine the effect of current timber management practices (in particular, the observance of buffer strip guidelines) on small "cold-water" lakes fish habitat. From the beginning, the transfer of nutrients from watershed to lake was a major concern of the study. Nutrient transfer directly from the forest edge was identified as a potentially significant pathway that could be affected by forest cutting to the shoreline, and a pathway which has received little study. In conjunction with the Ontario experiment, the Department of Fisheries and Oceans funded the measurement of nutrient inputs to lakes via litterfall from the forest edge.

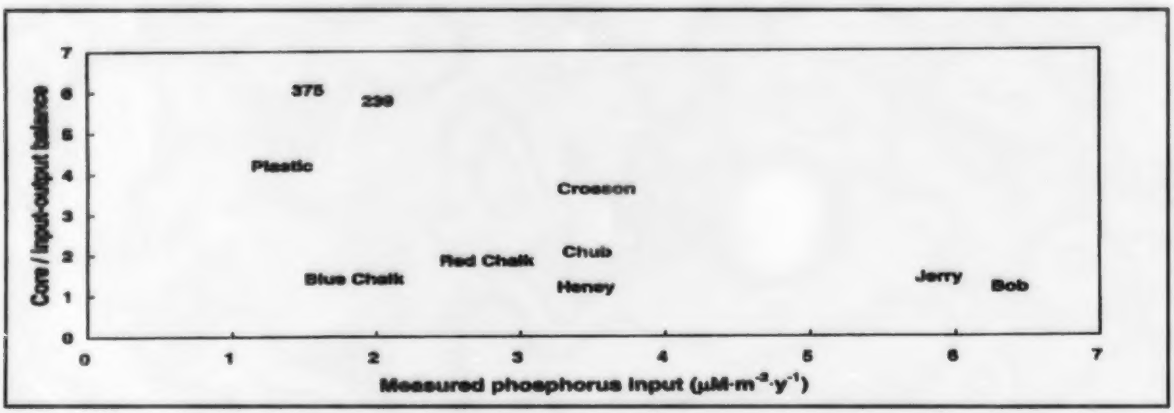

Figure 1. The ratio of phosphorus sedimentation rates as determined by core analysis versus input–output balances as a function of phosphorus inputs measured in influent streams and precipitation. Southeastern Ontario data from Evans and Rigler (1980), Evans and Rigler (1983), and Dillon and Evans (1993). Experimental Lakes Area data from Bayley (1992), Kipphut (1978), Campbell (1994), and P. Wilkinson (personal communication).

Lake 239 in northwestern Ontario was chosen as a site for a preliminary study because of the long record of nutrient balances and productivity measurements associated with it.

Site Description and Methods

Lake 239 is a 54.3 ha lake set in a 393 ha watershed. The watershed lies on a granodiorite batholith which forms rolling, glacially carved topography with local relief of generally less than 50 m. Nutrient fluxes in the three influent streams and the outflow have been measured continuously since 1970 (Schindler et al. 1976; Bayley et al. 1992).

The surrounding forest was described by Brunskill and Schindler (1971) as "typical ... boreal forest subclimax," and is dominated by tall jack pine (*Pinus banksianna*) and white spruce (*Picea glauca*). It was burned to a large extent by successive forest fires in 1974 and 1980, and now is in an early stage of regeneration. Except for an old growth stand adjacent to the field camp, young jack pine and paper birch (*Betula papyrifera*) now dominate the forest in the Lake 239 watershed.

One litterfall sampling site (WD) was adjacent to the remaining stand of old growth jack pine forest; two (WG, WH) were offshore from 15-year-old jack pine and paper birch dominated forest, growing back after the 1980 fire (Fig. 2). The forest at site WD was open canopy, dominated by 100-year-old, 25-m tall jack pine, with some paper birch and an open understory dominated near the shore by alder (*Alnus* sp.) and young paper birch. At the water's edge were discontinuous clumps of leather leaf (*Chamaedaphne* sp.) and bog laurel (*Myrica gale*). At the start of sampling in spring, the water's edge was adjacent to, and partly under, the clumps of leather leaf. As the season progressed, lake levels dropped by 0.2 m, and a 0.5- to 1-m wide beach of small boulders, cobbles, and pebbles was exposed between the dense vegetation and the water's edge. Site WG and WH were adjacent to a dense stand of young, 4–6-m tall jack pine. Small stands dominated by paper birch occurred among the jack pine, one being near to site WG. At the shore, alder and small birch were common, and leather leaf was dense and almost continuous. Leather leaf overreached the water until at least mid-summer, by which time the lake had receded to leave a fraction of a metre of pebble and pebble beach between the leather leaf and the water. All 3 sites were along the west shore of the lake, thus sharing the same general sun angle and wind events.

Near shore, litterfall sampling transects were grouped in 3 lines at WD, and 2 lines each at WG and WH. Transects at each site were approximately 10 m apart, with pans at 1, 3, and 9 m from the water's edge. At sites WD and WH, pairs of pans were floated on rafts at 30 and 90 m distance from the water's edge. An additional float with 2 pans and a rain collector was set near the center of the lake at 250 m from shore (Fig. 2, site CB). In midsummer a pan was added at 0 m (the water's edge) on each transect. Distances were measured from the water's edge at high lake level, as it occurred immediately after breakup in early May. As the lake level dropped 0.2 m through the summer, pans nominally at 1 m were 0.1–0.6 m from the water's edge by late summer, and pans set at 0 m were actually on shore.

Litterfall collection pans were 27.2 cm in diameter (0.058 m^2) by 9 cm deep, cut from the bottoms of white plastic pails. Near to shore, pans were set on platforms on wooden poles driven into the lake bottom. Offshore pans were set on long, narrow rafts constructed of two 5.5-m long 1 × 6 boards (2.5 × 15 cm) set on edge 75 cm apart, with small platforms near the center holding two pans 25 cm above water level. To increase their mass, the rafts were weighted with cinder blocks hung from the corners down into the water. The rafts rode perpendicular to wave crests, holding the shallow pans just above maximum wave height, and sitting stable enough in rough water that litter loss due to rocking was rare.

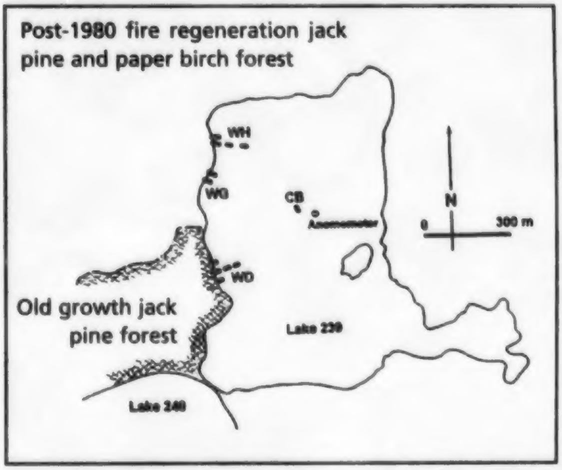

Figure 2. Lake 239, showing locations of litterfall sampling transects. The northern limit of the remaining stand of old growth forest is indicated by the dashed line.

One litre of dilute KCl solution (KCl added to distilled, deionized water to 100 $\mu S \cdot cm^{-1}$ conductivity) was poured into to each pan when it was set on the lake. The period of deployment was 2–4 days. Upon retrieval, larger objects in each pan were briefly described in the field, and the contents of each pan were emptied into separate 2 L Nalgene bottles for transport to the field laboratory. Aberrant sample volumes and conductivities were noted, and if there was contamination by the splash of lake water into the pans, the samples were discarded.

Samples were filtered through a preignited Whatman 1.2-μm glass fiber filter. A 125-mL volume of filtrate was refrigerated and sent to the Freshwater Institute in Winnipeg for determination of dissolved phosphorus (DP), nitrogen (N), and organic carbon (OC). Analyses were completed within a few days to 2 weeks after sampling.

Filters were placed flat in covered petri dishes and frozen for later visual description and chemical analysis. Filters were examined under 10 × 50 power binocular microscopes. Care was taken to minimize possible contamination or loss of material that would affect later analysis for nutrient content. Where possible, particles were identified with emphasis on discriminating between particles of lake origin versus particles of terrestrial or other non-lake origin. Plant material was identified to species and part (i.e., bark, leaf, pollen, seed, etc.) where possible. Insects were generally identified to family, although in some cases (especially Dipterans) this was not sufficient to establish provenance.

After visual examination, filters were analyzed for particulate phosphorus (PP); a subsample was also analyzed for N and OC. Filters destined for only PP analysis were submitted to the lab immediately after visual inspection. Filters intended for PP, N, and OC analysis were first homogenized and split. Homogenization was accomplished by immersing the sample in a small quantity of water and grinding with a Junke and Kunkel high-speed tissue grinder. The resultant slurry was refiltered, the filter split, and one half was submitted for P analysis, the other for N and OC analysis. The filtrate was analyzed for P, N, and OC, and the weights in the filtrate and on the filter were summed and doubled to give total PP, N, and OC.

Only P will be discussed in this paper. Dissolved phosphorus was analyzed with a Technicon colorimetric autoanalyzer according to the method of Stainton et al. (1977). Particulate samples on filters were ignited at 550°C, digested in 2 mL 1 M HCl with 10 mL distilled water, and then analyzed for DP.

Bulk precipitation chemistry was measured by ELA field laboratory staff at a site on Lake 240, about 1.5 km south of the litterfall sample sites. This site has been used to measure precipitation chemistry since the early 1970s, and is located on an island to minimize local litterfall inputs. It is covered with a 1-mm mesh screen to exclude insects and other potential large anomalous inputs. Samples are retrieved during and after large rainfall events, or at the end of any period during which greater than 10 mm rainfall has fallen.

The total P catch for each pan-deployment was corrected to exclude P associated with more widespread precipitation events. This was accomplished by subtracting P measured in the bulk precipitation collector for the period of deployment. Deposition for the whole lake was calculated separately for three 2-month periods: May–June, July–August, and September–October. Mean P deposition rates (after subtracting P in bulk precipitation) for pans at 0, 1, 3, 9, 30, and 30–250 m were calculated for each 2-month period. Phosphorus deposition was calculated for zones 0–1, 1–3, 3–9, and 9–30 m distant from shore by multiplying the mean deposition rate for pan deployments on each side of the zone by the duration of the period, and by the surface area of that zone around the whole lake. The mean P deposition rate for all pan deployments at 30–250 m was multiplied by the duration, and by the area of lake surface greater than 30 m from shore. The total deposition for each zone, and for the main central area of the lake, were summed to estimate the total P deposition for each of the 2-month periods. This calculation was repeated separately using data first from sites WB and CB, and then from sites WG, WH, and CB. This method developed separate estimates of P deposition over the whole lake as if it were surrounded entirely by old growth (WD, CB) or regeneration (WG, WH, CB) jack pine forest, respectively.

Table 1. Rate of phosphorus deposition per pan deployment ($\mu M \cdot m^{-2} \cdot d^{-1}$)

	Number of samples	Mean	Standard deviation	Dissolved	Particulate
May–June	70	6.3	12.0	1.3	5.0
July–August	67	3.5	5.2	1.2	2.3
September–October	147	1.8	6.0	0.5	1.3
Open water season	284	3.3	8.0	0.9	2.4

Results

Technique

Pans were tested for leaching of P by setting out 1 L of KCl solution in each of six pans, sealed in plastic sample bags for 2 days. The analysis showed no detectable P in the KCl solution (detection limit: 0.03 $\mu M \cdot L^{-1}$). In addition, every sample period included 1 or 2 blanks. Complete field procedures were carried out, except that pans set out in the field as blanks were lidded, and wrapped in plastic sample bags. The initial sample procedure was conducted until late June, and produced blank values with a mean of 0.17 $\mu M \cdot L^{-1}$ P. More careful field technique resulted in blanks for all of July to October, ranging from 0.03–0.16 $\mu M \cdot L^{-1}$ P and with a mean of 0.07 $\mu M \cdot L^{-1}$ P. Since the test for leaching showed no measurable P, it can be presumed that all P measured in blanks resulted from contamination during field and filtration procedures.

Phosphorus in Collectors

A subset of 284 samples have been analyzed for dissolved and PP. The mean total P measured per pan deployment was 3.3 $\mu M \cdot m^{-2} \cdot d^{-1}$ [standard deviation (s) = 8.0 $\mu M \cdot m^{-2} \cdot d^{-1}$]. Corresponding seasonal data are tabulated in Table 1. In every case, the standard deviation is greater than the mean. This is not surprising, considering the frequency of occurrence of coarse particulate matter in the samples. Frequently, most of the P in a sample was contributed by one or two insects, or other relatively large particles.

Particulate phosphorus (here defined as material held on a 1.2-μm glass fiber filter) made up 72% of all P measured; it dominated in all seasons. The high proportion of PP in May and June matches the period of peak intensity of jack pine pollen dispersal. High PP values in both spring and summer may also be associated with the peak seasons of insect productivity in the forest. Both pollen and insects were found in pans throughout the lake, though most large insects were caught in pans near shore. On two occasions, flying carpenter ants swarmed and were found scattered every few metres over the whole lake surface. Interestingly, no peak of litterfall P was observed in the autumn leaf fall period, although Site WG was near a deciduous stand (paper birch). Although a few deployments caught abundant leaves, especially leather leaf, they were found almost exclusively in pans within 1 m of the water's edge. Insects continued to be found abundantly in pans throughout the autumn, but with very small chironomids dominating to a greater extent than earlier in the year. A preliminary estimate of the provenance of coarse material collected has been made (Table 2).

Highest total and particulate P deposition rates were observed within 9 m of shore (Fig. 3) and appeared to reach an asymptote by 30 m (by 3 m in autumn). This trend was slightly less pronounced in the dissolved fraction. Differences between sites in the shapes of these distributions were noted, and are discussed below.

Table 2. Provenance of material found in litterfall collectors

Origin	Estimated volume (mm^3)
Originating in forest, streams	
Insects	90
Coarse plant material (not including pollen)	545
Originating in lake	
Insects, spiders	48
Undetermined origin	
Insects, spiders	28

Note: Volumes reported are based on counts and estimates of major dimensions as observed under a binocular microscope. (Subsample size: 111 filters, all seasons represented approximately equally.)

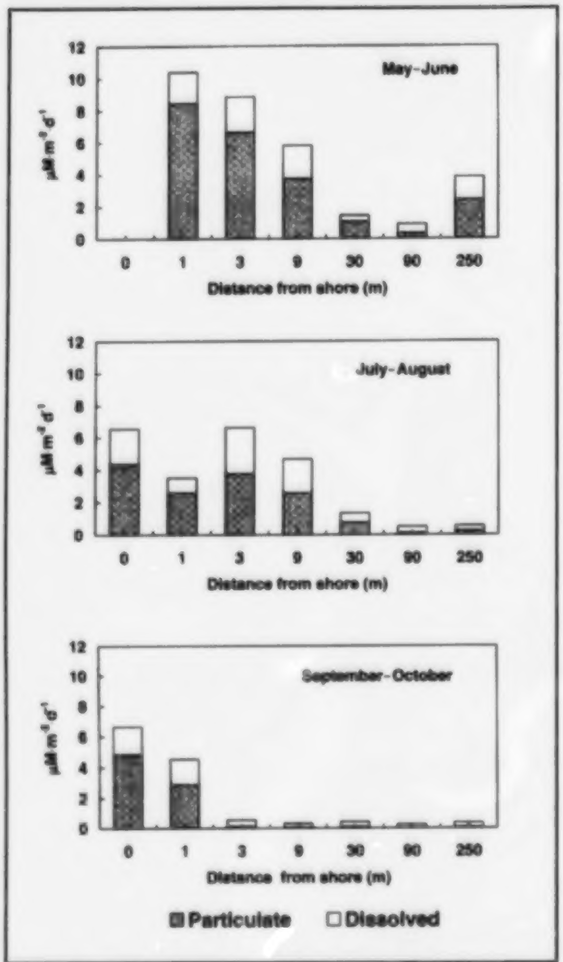

Figure 3. Phosphorus deposition as a function of distance from shore, shown seasonally.

Phosphorus in Biota Associated with Litterfall

A small sample of insect and plant parts were analyzed for concentration and weight of P (Table 3). Insects analyzed ranged in P concentration from 3 to 12 mg·g^{-1}, and contained a mean of 9 mg·g^{-1} (about 1% dry weight). Highest values were in the Dipterans, which dominated the litterfall catch. Hymenoptera, at the low end of the range, fell on the lake during at least two widespread flying ant events.

Broad leaves and needles contained the lowest P concentrations measured, one and two orders of magnitude lower, respectively, than insects. Phosphorus in leaves and needles was compared between live, green material and dried, brown

Table 3. Phosphorus concentrations (mg·g^{-1}) in some biota associated with litterfall into Lake 239. Dry weights were obtained by either freeze-drying or oven-drying overnight at 100°C. Phosphorus concentrations are per unit dry weight.

	Mean	Standard deviation	Sample size
Insects	8.9	3.4	13
Flies	10.6	2.3	9
Ants	3.7	0.4	2
Leaf and plant hoppers	6.2	1.2	2
Plant parts	2.0	1.8	18
Dry leaves	0.9	0.3	3
Green leaves	1.1	0.3	3
Dry needles	0.2	0.03	2
Green needles	0.8	0.01	2
Jack pine pollen	4.4	0.06	2
Jack pine flowers	2.2	0.6	2
Seeds	3.4	1.9	6

foliage, the latter being the usual form of foliage found falling onto the lake surface. Jack pine and black spruce needles, which were found in samples throughout the open water season, apparently withdraw about three-quarters of initial P before allowing needles to fall. In contrast, alder and leather leaf (two of the main leaf sources to Lake 239 in autumn) did not seem to withdraw P before shedding leaves. Plant parts associated with reproduction (pollen and seeds) were much higher in P than foliage, and were equivalent to the lower end of the range of P concentration in insects. Jack pine pollen, with a P concentration of 4.4 mg·g^{-1}, comprised a major part of litterfall in June and July.

Discussion

Collector Efficiency

Lewis (1983) compared the collection efficiencies of wet and dry surface collectors, and found them significantly different in their ability to catch metal cations and chloride, but not in trapping orthophosphate, soluble organic phosphorus, or particles in general. He attributed the greater collection efficiency of metal cations and chloride in wet traps to the dissolution of aerosols, a process which does not occur in dry traps. This process apparently does not affect P collection efficiency. Cole et al. (1990), however, in a 3-day deployment, found that shallow pan collectors containing water caught 10 times more P than dry pans. The contradictory results may be due

to differences in the type of collector. Lewis compared a funnel-shaped dry-fall trap to a 0.2 m^2, 12-cm deep box filled with 7.5 cm of distilled water. In contrast, Cole et al. compared 0.057 m^2, 7-cm deep collector boxes to wet collectors filled with 3 cm of deionized water. It is possible that local turbulence due to the funnel shape makes up in particle trapping efficiency what is lost in lacking a wet surface. This would account for Lewis' similarity in P collection efficiency between traps. Perhaps more importantly, Lewis' collectors were located in a wind-shielded clearing, while Cole et al. tested their collectors on the surface of Mirror Lake, exposed to a maximum over-water fetch of about 500 m. Under the latter conditions there is a greater potential for particles to be blown from a dry collector between samplings. Part of the catch reported by Cole et al. and the catch in pans on Lake 239 was free-flying insects that were trapped on the water surface. These would generally not be held in a dry collector.

A concern regarding meteorological collectors is elevation of the collecting surface. Catch efficiency can decrease with increasing height above the local surface. To minimize the frequency of contamination of the sample by waves splashing lake water into the pans, stands held the pan rim elevation at 0.25 m above water level. In a test of collector efficiency at elevations from 0.15 to 0.6 m above the ground, no significant difference was found in catch efficiency. This may have been due in part to the inclusion in the catch of free-flying insects in addition to particles transported solely by air motion.

Mean Rate of Litterfall Deposition on Lake 239

Mean rates of litterfall P deposition for the open water period are shown in Table 4. Because most of the shore of Lake 239 is cloaked in regeneration jack pine forest, the best estimate of P deposition for the whole lake was 1.2 $\mu M \cdot m^{-2} \cdot d^{-1}$. Litter from the forest edge deposited during the period of pan deployment (Table 4) was responsible for one and one-half times as much P deposition as was measured in the bulk precipitation collector. Litterfall P for the whole open water season was about 200 $\mu M \cdot m^{-2} \cdot y^{-1}$. This represents 7–20% of the range of annual P inputs in precipitation plus runoff measured over the period 1970–89 (Table 5).

Overall, the old growth jack pine forest appears to have produced essentially the same mean whole lake litterfall P as the adjacent regeneration jack pine forest: 1.3 $\mu M \cdot m^{-2} \cdot d^{-1}$ for old growth, as compared with 1.2 $\mu M \cdot m^{-2} \cdot d^{-1}$ for regeneration. Data from the same center lake station is used in both calculations.

Table 4. Mean rates of litterfall P deposition ($\mu M \cdot m^{-2} \cdot d^{-1}$) for the 1995 open water period, on Lake 239, assuming either old growth or regeneration jack pine forested shorelines

	Rainfall	Old growth forest	Regeneration forest
May–June	1.9	3.1	2.7
July–August	0.5	1.0	1.1
September–October	0.3	0.2	0.2
Open water season	0.8	1.3	1.2

Note: The samples on which the estimate is based represent pan deployments of less than 10 days per month. Phosphorus in rainfall was measured independently in a bulk precipitation collector, and is reported here for only the periods included in the litterfall study.

Table 5. Comparison of phosphorus inputs ($\mu M \cdot m^{-2} \cdot y^{-1}$) to Lake 239 by precipitation (as measured in a bulk precipitation sampler), watershed runoff, and litterfall

	Minimum annual	Maximum annual
Precipitation	402	1716
Runoff	621	1241
Litterfall	204	204

The old growth (WD) and regeneration (WG, WH) litterfall study sites are as close to each other as to the center of the lake. Therefore, it is inevitable that their separate effects will be unmeasurable beyond some undetermined distance from shore. The mean rate of P litterfall deposition per pan deployment within 9 m of the water's edge was 42% higher for old growth than for regeneration forest: 5.1 $\mu M \cdot m^{-2} \cdot d^{-1}$ ($s = 11.1$, $n = 83$) for site WD, versus 3.6 $\mu M \cdot m^{-2} \cdot d^{-1}$ ($s = 7.3$, $n = 111$) for sites WG and WH. In contrast, the mean rate at 30–90 m from shore adjacent to old growth forest was less than half that associated with regeneration forest: 0.22 $\mu M \cdot m^{-2} \cdot d^{-1}$ ($s = 0.33$, $n = 31$) for site WD, versus 0.48 $\mu M \cdot m^{-2} \cdot d^{-1}$ ($s = 1.12$, $n = 31$) for sites WG and WH. The very high standard deviations (twice the mean) make any deduction tentative, but the smaller difference between old growth and regeneration for whole lake rates than for near shore rates suggests that, while overall production of litterfall P at the forest edge is similar between forest types, a higher proportion of litterfall P from the old growth forest falls nearer to shore. This could possibly be explained by the relative wind-shelter effect of the much higher old growth forest. It may also reflect

differences between sites in the relative fraction of litterfall comprised of jack pine pollen or other widely dispersed material. Such differences are, however, speculative.

Reported P deposition in litterfall for several lakes is shown in Table 6. Cole et al. (1990, Mirror Lake, New Hampshire, U.S.A.), Kowalczewski and Ryback (1981, Lake Warniak, Poland), and this study used water in floating pans as a collecting surface. This study follows the Mirror Lake study, with a similar design and period of deployment of individual collectors. In the Lake Warniak study, taller traps containing water and formalin as a preservative were sampled only monthly. They were deployed on two perpendicular transects at equal distances across the lake, with no traps nearer to shore than about 50 m. The Lake Warniak sampling design precludes determining if deposition is dependent on distance from shore, unless it continues to be significant over a Near shore zone wider than 50 m. In both Mirror Lake and this study, the curve of deposition rate versus distance from shore reaches its asymptote between 10–30 m from the water's edge. Thus, the P deposition rate reported for Lake Warniak (14 $\mu M \cdot m^{-2} \cdot d^{-1}$) would be conservative as all collection was done under the asymptote. In both the Lake Warniak and Mirror Lake studies, the reported deposition rates include traditionally measured bulk precipitation. The rates shown in Table 4 for Lake 239 were calculated by subtracting P measured in the ELA bulk precipitation collector. For comparison with other studies, P deposition rates reported for Lake 239 in Table 6 have been adjusted to include P in precipitation.

About three-quarters of the drainage area of Lake Warniak is meadow and farmland. Mirror Lake is surrounded entirely by mature, mixed hardwoods and conifers. Lake 239 is surrounded by jack pine forest. Because the sampling design is similar in the Mirror Lake and Lake 239 studies, it seems likely that the large difference in deposition rate per unit shoreline length is real. The difference is probably due to some combination of differences in forest productivity, or in slope from the forest top to the water's edge. The agricultural land around Lake Warniak may be more productive as a litterfall source than either deciduous or coniferous forest. It is also possible, however, that collector design (i.e., much larger height-to-width ratio, formalin in the sampling solution) and deployment pattern in the Lake Warniak study could explain the much greater litterfall on a per unit shoreline length basis. Because of the significance of free-flying insects to total P deposition in all studies, the possibility that some traps attract insects, or that different trap designs attract insects differently, cannot be dismissed.

Significance of Insects to Litterfall

In all three studies using wet pan style traps, arthropods comprised a significant proportion of total catch. In Lake Warniak, invertebrates (mostly insects) comprised 38% of P measured. Since trap water was collected only monthly, it seems likely that leaching would have moved some P from the invertebrate fraction to dissolved form, and therefore P measured would be a minimum estimate. At Mirror Lake, arthropods accounted for up to 30% of the total number of particles caught. Because of higher P concentrations in arthropods than in plants, the former probably comprised a larger fraction of the P measured than the plant portion. Both Kowalczewski and Ryback, and Cole et al. reported that the vast majority of insects in litter collectors were of terrestrial origin. In this study, less than about 20% of the volume of material collected (excluding the sometimes very abundant pollen, Table 2) was comprised of arthropods (predominantly insects). Arthropods tended to be smaller than plant fragments, and so arthropods may have made up a numeric fraction of litterfall particles as large as that observed in at Mirror Lake, and therefore accounted for a similar fraction of P input. However, almost 30% (by volume) of arthropods counted in this study had lacustrine life stages, and another 17% were of undetermined origin. This is a higher fraction of arthropods originating in the lake than at either Lake Warniak or Mirror Lake. This reflects an unestablished fraction of total P in litterfall at Lake 239 that cannot be considered an input to the lake.

Rau (1976, Findley Lake, Washington State, U.S.A.), Gasith and Hasler (1976, Wingra Lake, Wisconsin, U.S.A.), and Hanlon (1981, Llyn Frongoch, Wales) deployed litterfall collectors designed to collect coarse material only, with open-mesh bottoms to pass rain water. In the Findley Lake study, insects were removed and not analyzed, and conifer needles accounted for 89% of material caught. At Lake Wingra, wire mesh traps caught exclusively leaves (>83%), fruits, and seeds. Hanlon at Llyn Frongoch observed only leaf litter (95%) and twigs in his conical nylon mesh traps.

The data reported in Table 6 were collected at different times of year, and are based on different periods between individual sample deployments and

Table 6. Comparison of phosphorus (P) deposition reported for several lakes. Data shown for Lake Warniak, Mirror Lake, and Lake 239 include P in precipitation. Data for Findley Lake, Llyn Frongoch, and Lake Wingra are for coarse particulate material only; for Llynn Frongoch and Lake Wingra, P concentration = 0.1% is assumed.

	Lake area (ha)	Shore length (km)	Season	Sample period (d)	P deposition per unit area ($\mu M{\cdot}m^{-2}{\cdot}d^{-1}$)	P deposition per unit shore length ($\mu M{\cdot}m^{-1}{\cdot}d^{-1}$)
Litterfall sampling in wet pans						
Mirror Lake	15	1.9	Late June to early September	49	12.0	950
Lake Warniak	38	2.7	June–September	92	14.0	1900
Lake 239	54	4.5	May–October	170	2.0	240
Litterfall sampling in dry baskets						
Findley Lake	11	1.4	June–October	117	0.8	63
Llyn Frongoch	7	1.4	October–December	64	0.5	26
Lake Wingra	130	5.0	Autumn	30 (est.)	2.9	760

Note: Sources for Mirror Lake = Cole, Caraco and Likens 1990; Lake Warniak = Kowalczewski and Ryback 1981; Lake 239 = (this study); Findley Lake = Rau 1976; Llyn Frongoch = Hanlon 1981; Lake Wingra = Gasith and Hasler 1976.

collections. For Lake Wingra and Llyn Frongoch, the data represent rates for a brief, intense burst of litterfall associated with autumn leaf fall in temperate deciduous forests. Both Gasith and Hasler, and Hanlon suggest that the total litterfall over the leaf-out period would have been 120% of that measured in the autumn. Mean P deposition rates for periods comparable to the others listed in Table 6 would be of the order of 0.3 $\mu M{\cdot}m^{-2}{\cdot}d^{-1}$ and 0.9 $\mu M{\cdot}m^{-2}{\cdot}d^{-1}$ (about 15 $\mu M{\cdot}m$ $shore^{-1}{\cdot}d^{-1}$ and 200 $\mu M{\cdot}m$ $shore^{-1}{\cdot}d^{-1}$) for Llyn Frongoch and Lake Wingra, respectively. Except for Lake Wingra, deposition rates per unit shore length measured in open-bottom style collectors are much lower than those measured in wet pan style collectors. Wet pan style collectors measure P in precipitation, which would wash through open bottom collectors. Moreover, rainwater washing through litter (especially if sampled at long intervals) may leach some P from the wetted mass, and effectively remove smaller particles such as pollen from the collector. The water surface in a wet pan is also more likely to trap insects landing on the surface. Traditionally, insects in precipitation have been considered as a nuisance in bulk precipitation samplers, which commonly employ a fine mesh screen designed to exclude them. However, they are the carrier of a considerable proportion, in the range of 20–40%, of total P transported from the forest edge onto the lake surface in the three studies designed to measure them.

The importance of insect P to total litterfall deposition has several implications. In terms of process, free-flying insects as a transport mechanism present different sampling problems than suspended particles. There is a serious concern that the type of collector may bias P deposition measurement by attracting insects. Chironomids are known for mating over strong landscape markers, and spiders seek out and use litter pans as habitat. Because they emerge from within the lake, chironomids are, in any case, not a net source of P. Live spiders were always removed from samples (very few were found dead). The net effect of spiders on litterfall collection was negative, because they fed on insects in the sample; partially eaten insects were commonly found in the litterfall collectors. The net overall effect of these methological concerns was not established in this study.

Conclusions

Litterfall (sometimes called local aeolian input or short-range atmospheric transport) is in large part driven by biological forces. The interpretation of pan-caught data may require different process considerations than other meteorological sampling. Biological productivity, whether of plants or animals, controls the mass available for transport. Biological activities (such as ant swarming, insect emergences, and mating events) may in themselves be considered transport events. In the case of such high-intensity, short-lived events, weather factors such as wind and rain are secondary agents of nutrient and elemental transport.

Only a subsample of data collected in the field season has been considered here, and only preliminary inferences can be drawn. The contribution of litterfall from the forest edge to the P balance of Lake 239 is significant. It is of the same order of magnitude as the traditionally measured input by summer precipitation. Litterfall P delivery, unlike precipitation P, is concentrated in the littoral zone. Litterfall P is also qualitatively quite different from precipitation P with respect to the greater proportion of high quality protein that litterfall brings into the food web at a potentially higher trophic level.

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Predicting the Importance of Boreal Forest Streams as Fish Habitat

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Abstract

Predicting the importance of boreal forest streams as fish habitat is a problem faced by fisheries biologists and managers of forestry operations. Multiple regression analysis was employed to determine the physical stream parameters that could be used to predict species richness in fish communities of boreal forest streams in Saskatchewan. The regression model with the highest coefficient of determination ($r^2 = 0.539$; $p < 0.05$) for predicting species richness had the $\log_{10}$ of the mean wet width of the stream as the only independent variable. The equation for the model was:

$$\textit{Species richness} = -0.157 + 3.618(\log_{10} \textit{wet width}).$$

The species richness model accounts for approximately 54% of the variation in the dependent variable. Habitat diversity and the position of a stream in the drainage network have been suggested in the literature as factors that may influence the species richness of stream fish communities. These factors should be considered in any further investigations of fish communities in boreal forest streams. Guidelines for sampling boreal forest stream fish communities are also described. These include sampling at baseflow, and sampling a stream length of at least 100 m. Brook sticklebacks were the most common fish species. Other common fish species included white sucker, pearl dace, longnose dace, and slimy sculpins.

Merkowsky, A. 1998. Predicting the importance of boreal forest streams as fish habitat. Pages 169–176 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

One of the problems facing fisheries biologists and managers of forestry operations is predicting the importance of streams as fish habitat. This is particularly a problem in the boreal forest in Saskatchewan, where the data describing the fish communities in streams is limited to a study conducted by Larson (1976). This is in contrast to mountainous regions, where the impact of forestry operations on fish habitat and the associated communities has been well documented (Chamberlin et al. 1991).

One result of Larson's (1976) study, was recommendations for implementing buffer strips to protect fish habitat from the impact of forestry operations in Saskatchewan. Part of these recommendations included criteria for determining the size of buffer strips to protect forest streams. These criteria include: stream order number, watershed area, and the presence or absence of commercial game fish populations. In recent years, concerns have been raised about the suitability of these criteria for determining buffer strip width. A second concern was that Larson's study only focused on one region of the boreal forest; the Cub Hills in east-central Saskatchewan (Fig. 1). Extrapolation of results from this area to the whole boreal forest was questionable, and so Saskatchewan Environment and Resource Management (SERM), in conjunction with Forestry Canada (Natural Resources Canada), conducted a study of fish habitat in streams of the boreal forest to address this concern.

The first objective of this study was to examine the criteria for determining the width of buffer strips for protecting lotic habitats, and for predicting the importance of streams as fish habitat. This would involve developing models for predicting the size of a stream and its species richness (i.e., the number of fish species present). The second objective was to determine the most likely fish species to occur in a particular segment of stream.

Materials and Methods

The study area (Fig. 1) is located in the commercial forest zone of Saskatchewan. This zone includes part or all of the following ecoregions: Churchill River Upland, Mid-Boreal Upland, Mid-Boreal Lowland, and the Boreal Transition (SERM 1995). The majority of the streams sampled in this study were located in the Mid-Boreal Upland ecoregion.

Between 1992 and 1994, data were collected from 51 stream sites, from two sources. Twenty-seven sites were located within six watershed systems: Broad Creek, Twoforks–Randall rivers, Bear River, Rossbell Creek, McDougal Creek, and the Wuchewun River. Data on the physical parameters and fish communities of an additional 24 sites were obtained from a concurrent study on the impact of forest harvesting on boreal forest streams (Gloutney and Merkowsky 1995). The sampling period during each year was from mid-May to mid-August. The four sites on the Wuchewun River and 14 of the sites from the forest harvesting project were only sampled on one date. All the other 33 sites were sampled on more than one occasion to examine the stability of fish communities between years. (This aspect of the fish communities in boreal forest streams will be addressed in a separate paper.) The selection of study sites was based on accessibility, the drainage basin area for the site, and the presence of a definable stream bank. The latter criterion eliminated beaver ponds as possible sites.

A number of physical parameters, categorized as either drainage basin or channel morphometry type, were determined for each site. The drainage basin parameters were: watershed area upstream of the sample site, stream order number, and segment link number (the sum of the first order streams that have joined to form the stream at the sample site) (Osborne and Wiley 1992). These parameters were determined from NTS (1:50 000 scale) topographic maps. Channel morphometry parameters were: bankfull width, wet width, depth, and volume. The bankfull width was used as an indicator of the bankfull discharge, which is the flow that fills the channel without overflowing onto the floodplain (Platts et al. 1983). Wet width was measured as the actual portion of the stream channel that contained water at the time of sampling.

At each stream site, 9 to 11 permanent transects were established on the stream in accordance with methods described by Platts et al. (1983). These transects were generally 10 m apart, though on some of the larger streams the transects were 15 or 20 m apart. Bankfull width, wet width, and depth were measured at each transect, following the methods of Platts et al. (1983). From these measurements, mean values for each of the channel morphometry parameters (including volume per 10 m of stream length) were calculated. For those sites that were sampled on more than one date, an overall mean was calculated from the different sample dates for use in the statistical analyses.

The values for the physical parameters from all sites were examined for normality using the Lilliefors Test (Norusis 1993). A normal distribution was approximated by a $\log_{10}$ transformation of the data; therefore, the transformed values for the physical

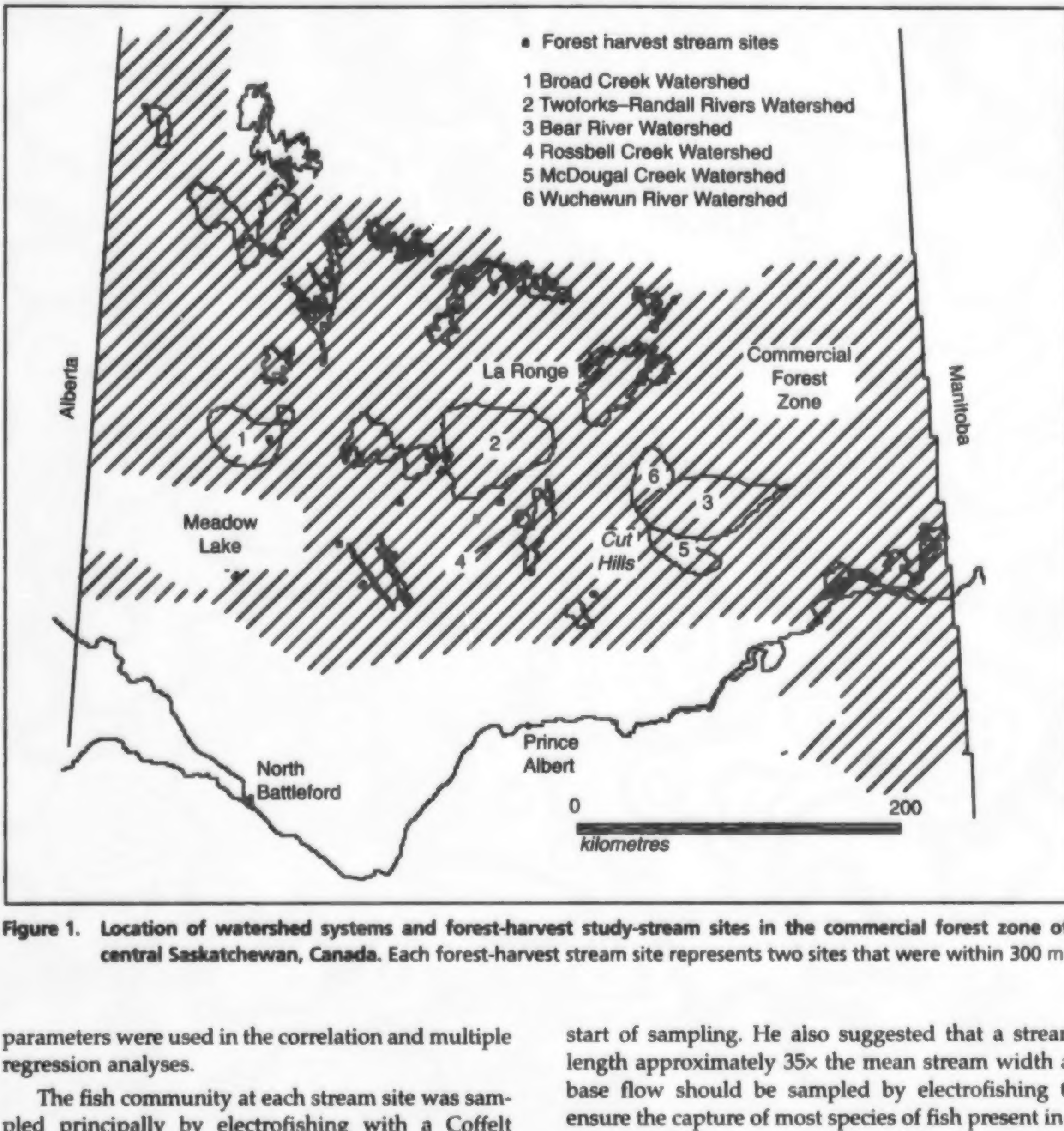

Figure 1. Location of watershed systems and forest-harvest study-stream sites in the commercial forest zone of central Saskatchewan, Canada. Each forest-harvest stream site represents two sites that were within 300 m.

parameters were used in the correlation and multiple regression analyses.

The fish community at each stream site was sampled principally by electrofishing with a Coffelt Mark 10 electrofisher. The area to be fished was blocked at the upstream and downstream ends with seine nets, and two or three passes were made through the site. After each pass, the species of fish were identified and counted.

The amount of effort required to sample the species richness in a stream was examined by Lyons (1992). He determined that the rate of increase in the cumulative number of species found generally began to slow down within 100 m (stream length) of the start of sampling. He also suggested that a stream length approximately 35× the mean stream width at base flow should be sampled by electrofishing to ensure the capture of most species of fish present in a stream. While this would be ideal, the sampling length in some of the larger streams was limited by the presence of deep pools. Consequently, these pools were sampled in 1993 with overnight sets of gill nets (38- and 78-mm mesh sizes). Generally, at least an 80-m length of stream was sampled at each site.

Electrofishing effectiveness varied between sites depending on the conductivity of the water and visibility, the latter being affected by the water color and depth. Susceptibility to the electroshock also varied

Table 1. Correlation coefficients between stream physical parameters ($\log_{10}$ transformed) and species richness for boreal forest streams in Saskatchewan. N = 51. All correlations were significant (p <0.05) except for depth and bankfull width.

Parameters[a]	BWdth	WWth	Depth	Volume	Order no.	Link no.	Wshed A
BWdth	1.000	–	–	–	–	–	–
WWth	0.953	1.000	–	–	–	–	–
Depth	0.204	0.380	1.000	–	–	–	–
Volume	0.763	0.879	0.768	1.000	–	–	–
Order no.	0.777	0.786	0.278	0.689	1.000	–	–
Link no.	0.831	0.839	0.346	0.759	0.935	1.000	–
Wshed A	0.911	0.888	0.325	0.790	0.832	0.917	1.000
Fish sp. no.	0.693	0.741	0.686	0.709	0.583	0.650	0.686

[a] BWdth = bankfull width, WWth = wet width, Depth = mean depth, Volume = volume per 10 m of stream length, Order no. = stream order number, Link no. = stream link number, Wshed A = watershed area, Fish sp. no. = species richness.

between species, with cyprinid species being generally more susceptible and sticklebacks the least. Habitat preference also influenced susceptibility; for example, sculpins were difficult to catch among the rocky substrates that they preferred. These differences in susceptibility may affect analyses of relative abundance of the different species, and some rare species may have been missed at some sites.

All statistical analyses were carried out using the SPSS for Windows Release 6.0 Base and Professional Statistics Modules. Pearson correlation coefficients were calculated to determine the association between the physical parameters. Based on the correlation analysis, stepwise multiple regression was then used to determine the relationships between the channel morphometry (dependent variable) parameter that had the highest correlation coefficient and the drainage basin parameters (independent variables). Similar procedures were used with the regression model for predicting species richness of the fish community at a site. Species richness was the dependent variable, while the independent variables ($\log_{10}$) in the multiple regression analysis were: watershed area, stream order number, segment link number, mean bankfull width, mean wet width, mean depth, and mean volume per 10 m of stream length.

Results

The Pearson correlation coefficients among the physical parameters are presented in Table 1. Watershed area and bankfull width had the highest correlation coefficient ($r = 0.911$) between the drainage basin and channel morphometry parameters. Among only the channel morphometry parameters, the highest correlation coefficient was between bankfull width and wet width ($r = 0.953$).

Based on the above correlation analysis, a regression model for predicting the size of a stream was developed using the mean bankfull width of a stream as the dependent variable and the drainage basin parameters as independent variables. The multiple regression analysis showed that watershed area was the only significant independent variable for predicting the mean bankfull width of a stream, by which:

$$\mathit{Log_{10}\ bankfull\ width} = -0.047 + 0.479(\log_{10} \mathit{watershed\ area})\ (p < 0.05).$$

The coefficient of determination for the regression model was $r^2 = 0.827$. Figure 2 presents a graph of the data with the slope and confidence intervals.

The physical parameter with the highest correlation coefficient ($r = 0.741$) with species richness was the mean wet width ($\log_{10}$ transformed) of the stream (Table 1). The drainage basin parameter with the highest correlation coefficient ($r = 0.686$) with species richness was watershed area ($\log_{10}$ transformed).

Regression analysis was conducted between species richness (as the dependent variable) and the physical parameters (as independent variables). The only significant independent parameter was the mean wet width of a stream, which is related to species richness by:

$$\mathit{Species\ richness} = -0.157 + 3.618(\log_{10} \mathit{wet\ width})\ (p < 0.05).$$

The coefficient of determination for the regression was $r^2 = 0.539$. Figure 3 presents a graph of the data showing the relationship between mean wet width ($\log_{10}$) and species richness for the sites.

Table 2. Number of occurrences of fish species in the 51 streams that were sampled

Species	Occurrence	Species	Occurrence
Brook stickleback	25	Northern pike	5
White sucker	18	Burbot	3
Pearl dace	14	Trout-perch	3
Longnose dace	9	Finescale dace	3
Sculpin	9	Brook trout	3
Fathead minnow	6	Yellow perch	2
Longnose sucker	5	Iowa darter	2

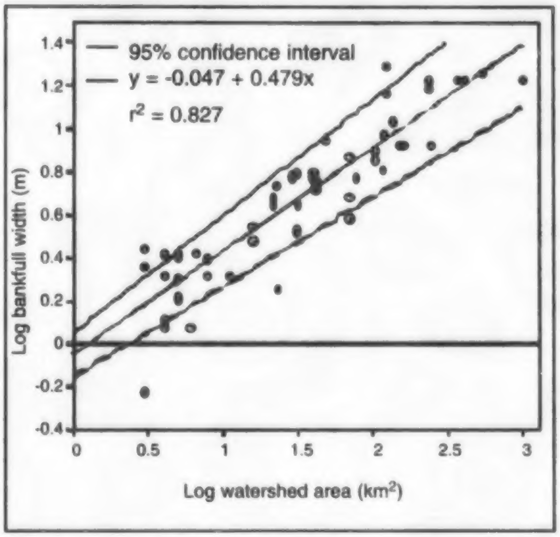

Figure 2. Regression model showing relationship between ($\log_{10}$) watershed area and ($\log_{10}$) mean bankfull width for boreal forest streams in the Mid-Boreal Upland ecoregion of Saskatchewan.

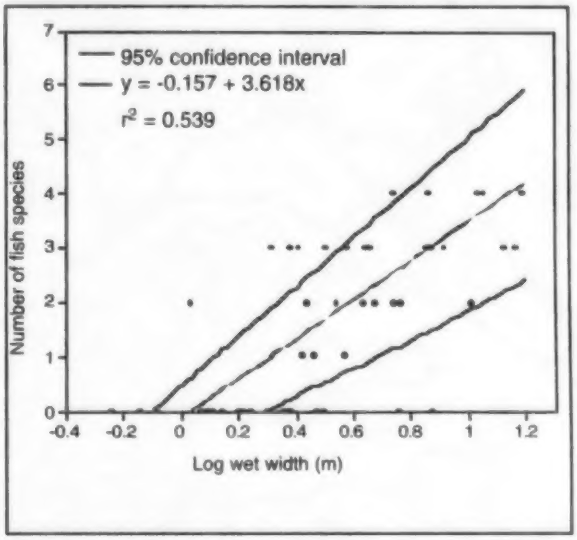

Figure 3. Regression model showing relationship between ($\log_{10}$) mean wet width and species richness for boreal forest streams in the Mid-Boreal Upland ecoregion of Saskatchewan.

The most common species of fish captured in the boreal forest streams was the brook stickleback (*Culaea inconstans*); it was found in 25 (49%) of the 51 stream sites that were sampled (Table 2). Other common (nine or more sites) species in descending order of occurrence were: white sucker (*Catostomus commersoni*), pearl dace (*Semotilus margarita*), longnose dace (*Rhinicthys cataractae*), and sculpin (*Cottus cognatus*). A total of 14 species were captured (Table 2).

Discussion

The above regression analyses have produced separate models for predicting stream size and species richness. One question that may be asked in comparing these models is: how well do they account for variation in the dependent variable? The coefficient of determination (r^2) for each model provides an answer to this question. It describes the percentage of variation in the dependent variable that is accounted for by the independent variable(s) (Sokal and Rohlf 1995). In the stream size model, it accounts for a higher percentage of the variation in the dependent variable than it does in the species richness model (83% vs. 54%, respectively).

What does the correlation coefficient tell us about the ability of the models to predict a value for the dependent variable? Prairie (1996) has shown that the predictive power of regression models with $r^2 < 0.65$ is low. Further, Prairie has suggested a measure called the resolution power (RP) index to evaluate the predictive power of regression models. This index uses the coefficient of determination to determine the number of distinct classes a model can predict, whereby:

no. of classes = $1.31/(1-r^2)^{1/2}$.

The value of the RP index for the stream size model is 3.2; for the species richness model, it is only 1.9.

Of the channel morphometry parameters, mean bankfull width ($\log_{10}$) had the highest correlation coefficient with any of the drainage basin parameters. The subsequent regression analysis showed that watershed area ($\log_{10}$) was the only significant independent variable for predicting the stream size as indicated by mean bankfull width ($\log_{10}$). The direct relationship between stream size and watershed area is well known (Leopold et al. 1964). Bankfull width is an indicator of bankfull discharge, which is the major channel-forming streamflow, having a recurrence interval of approximately 1.5 years (Leopold et al. 1964). Thus, bankfull width will not change appreciably over time unless there are long-term changes in bankfull discharge. By contrast, the other channel morphometry parameters (mean wet width, mean depth, and mean volume) depend on the discharge at the time of sampling. Discharge in boreal forest streams usually exhibits a peak in the spring and gradually decreases through the summer and fall to a winter low. However, this "regular" pattern may be altered by extremes in flow that are the result of extremes in precipitation (i.e., drought or flood). In such extremes, either of the channel morphometry parameters will be unreliable indicators of stream size.

From the RP index for the regression model for stream size, predictions may be divided into three categories. A boreal forest stream in Saskatchewan with a watershed area of 10 km^2 would have a bankfull width of approximately 2.6 m; streams with watershed areas of 100 and 500 km^2 would have bankfull widths of 8.1 and 17.5 m, respectively. One advantage of using watershed area is that it can be easily determined from topographical maps. Thus, a reliable estimate of stream size can be made without the expense of a field trip. This may be particularly useful in remote, hard-to-access locations.

The only significant independent variable for predicting species richness was mean wet width ($\log_{10}$). The model showed that, as the size of the stream increased, species richness also increased. The species–area relationship has been widely discussed in the ecological literature for many types of organisms (Connor and McCoy 1979). Angermeier and Schlosser (1989) have investigated some of the mechanisms of this relationship in stream fish populations. They found that species richness increased as the size of the sample area increased. The larger the sample area, the more individuals that are sampled, and thus the likelihood that more species are encountered. However, their analysis of the data also indicated that habitat diversity was also a factor affecting species richness. This suggests that, in segments of streams where the habitat is more diverse, species richness may be greater. Larson (1976) observed an increase in species richness with an increase in stream size as indicated by increasing stream order number. For the Mossy River system, streams of first, second, third, and fourth order had 4, 6, 9, and 12 species of fish, respectively. However, it was not clear if the data were the average number of species per stream or the total number of species found in streams of the respective order number.

Discharge is one factor that should be considered when evaluating species richness in streams. In this study, though the sampling period each year was from mid- May to mid-August, extremes in discharge did occasionally occur; these changes may have affected the sampling of the fish community in some streams. This is because of difficulties in sampling and because of changes in the distribution of fish at extremes in flow. High flows close to bankfull discharge make it difficult to see the fish, as the water is faster and is usually more turbulent. Shocked fish are also swept away by the current, making it more difficult to capture fish. Both very high and very low flows will force fish into refuges (Grossman et al. 1990) resulting in non-representative fish distributions during sampling. While most streams were sampled at base flows, as recommended by Lyons (1992), there were some exceptions. Figure 3 shows that there were four stream sites without fish that had mean wet widths ($\log_{10}$) greater than 0.4 (2.5 m). For three of those stream sites, the absence of fish could be associated with extremes in flow: two of them were at very low flows, while the other had evidence of recent high flows. These extreme events may have resulted in few or no fish at the site during sampling. The fourth site that produced no fish was one of the paired sites in the forest harvest study. The second site in this pair showed only a few (three) brook stickleback; thus, from the low numbers of fish captured, it is possible that fish were missed at the other site.

Another effect of sampling technique on the species richness model that should be considered is fish species vulnerability to the methods used. Because the actual length of a stream that could be sampled in some of the larger streams was constrained by deep pools, more species may have been captured if greater lengths of stream could have been

sampled, as suggested by Lyons (1992). Therefore, the predictions for species richness for the larger streams should be viewed with caution. The gill nets that were placed in some of the larger streams in 1993 did not capture any fish species that had not already been captured by electrofishing. However, the nets would only have targeted larger fish species; smaller fish species that only occupied deep pools would have escaped capture.

Another factor that may affect species richness in streams is a stream's position in the drainage network. Osborne and Wiley (1992) found that tributaries joining the main channel of a drainage network will have more fish species than similar-sized streams that are at the headwaters of the drainage network. They suggested that this is because main channel tributaries have a larger source of fish species available to them than do headwaters.

The predictive power of the species richness model is low. The RP index indicated that the model produces two classes: streams with fish, and streams that are not likely to have fish present. Examining the data in Figure 3, we observe that most streams with a $\log_{10}$ mean wet width value greater than 0.3 (2.0 m) had fish present, while only one stream with a mean wet width ($\log_{10}$) below 0.3 had any fish. This stream was sampled in consecutive years, and on both occasions, brook stickleback and pearl dace were captured, thus the presence of fish in this stream does not appear to be a chance occurrence.

Brook stickleback was the most common species of fish found in the streams. Larson (1976) found similar results in his survey of streams in the Cub Hills. White sucker, pearl dace, and longnose dace were other species that were found to be relatively common in this study and in Larson's. The occurrence of game fish species was relatively low at the sites in this study. This suggests that the presence or absence of game fish in a stream is not a reliable criterion for determining buffer width size. Rather, the presence of fish within the system would be more appropriate.

The coefficients for both of the above regression models will vary according to the geomorphology and climate of the watershed. As the streams that were sampled in this study were nearly all within the Mid-Boreal Upland ecoregion, this model would be most applicable to similar ecoregions. Similar observations have been made when considering other predictive models. For example, Fausch et al. (1988), in reviewing models for predicting standing crop of fish populations, observed that most models were most successful in their predictions when applied to areas that were within the same ecoregion. They also suggested that these predictive models be based on at least 20 sample sites. Both regression models meet these criteria.

This study is the first step toward determining the importance of boreal forest streams as fish habitat. The species richness model shows that streams smaller than 2 m (mean wet width) are not likely to have fish present. The model also shows that, as streams increase in size, they are likely to have more species of fish. However, with only a moderate coefficient of determination (r^2 = 0.539), other factors must also influence species richness. Thus the influence of habitat diversity, the position of a stream in the drainage network, and other factors should be investigated to refine the species richness model.

Refinement of the model can benefit from the following sampling guidelines. Sampling should be restricted to periods when base flow is occurring; sampling at high and very low flows should be done only to compare the effect of these extremes on the fish community present. The length of stream sampled should be at least 100 m, and alternative methods should be used to sample larger streams with pools that are too deep for backpack electrofishing. Suitable alternative methods include electrofishing from a boat, using minnow traps, and using fine (3.8 cm or smaller) mesh nets.

These models will help fisheries biologists and managers of forestry operations determine the appropriate conservation management practices for protecting streams from the impact of forestry operations. Even though the present model indicates that streams smaller than 2 m in mean wet width are not likely to have fish present, they still require protection, because inputs of organic debris or sediments may be carried downstream to those stream sections that do have fish.

The riparian zone adjacent to streams and lakes has several connections to the aquatic system and serves as a source of food, cover (i.e., large organic debris), and shade. This zone is also an important component of the terrestrial system. The use (size) of riparian buffer strips, therefore, must not only consider the protection of aquatic habitats, but the protection of plants and animals that are dependent on this portion of the forest.

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Problems, Practical and Psychological, Using Stable Carbon Isotope Ratios for Discerning Utilization of Allochthonous Forest Detritus by Stream Fauna: a Tale of Two Studies

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Abstract

A review of the literature has suggested that the capability of stable carbon isotope analysis for quantifying the incorporation of terrestrial detritus into stream fauna may be low. Here, I contrast two studies that were not part of my initial review, to provide further evidence for the inherent problems with stable carbon isotope ($\delta^{13}C$) analysis, which must be addressed and surmounted before this isotopic technique can be widely recommended for use in the research and management of land–lotic ecotones. In addition to the practical problems, the differing psychological perspectives demonstrated in the two studies are highlighted: the data interpretation phenomenology of Mize versus the data confirmation biases shown in the study led by Doucett, which provide insight as to how science is approached in general.

Introduction

It is assumed that the stable carbon isotope ratios (^{12}C:^{13}C expressed as $\delta^{13}C$ in relation to the isotopic standard in parts per thousand) of terrestrial plants differ from those of aquatic autotrophs, and these isotopic differences have been used for tracing energy flow pathways in stream ecosystems (Rounick and Winterbourn 1986). This has led to the suggestion that $\delta^{13}C$ analysis can be used as a simple biomonitor for gauging the aquatic effects of riparian deforestation (Winterbourn and Rounick 1985; Rounick and Winterbourn 1986). Recently, I reviewed the extant literature on stable carbon isotope ratios for streams and concluded that, contrary to the early optimistic beliefs, the great variability in attached algal $\delta^{13}C$ may often preclude use of stable isotope analysis for identifying carbon pathways in many of these systems (France 1995a). Because of this, I further concluded that the utility of stable carbon isotope analysis for understanding anthropogenic alterations to the carbon budget of streams is presently minimal. Despite these criticisms, however, some researchers have continued to uncritically promote $\delta^{13}C$ analysis as an ideal tool for use in applied studies concerning timber harvesting issues (Doucett et al. 1996).

In this paper I will highlight some of the misconceptions behind this continued "confirmation bias" (sensu Loehle 1987) in order to prevent these erroneous ideas from "muddling along in a plausible but unconfirmed state" (Loehle 1987) and thereby

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resulting in a weakening of our final scientific product (DeMelo et al. 1992). I will do this, not by reiterating my previous criticisms based on a review of worldwide data (France 1995a), but rather through a study-specific approach that criticizes and contrasts Doucett et al.'s (1996) confirmation biases with both the phenomenological data interpretations demonstrated in an unpublished thesis by Mize (1993) that has recently come to my attention, as well by my own appraisals.

Aquatic consumers rely more upon attached algae than vascular macrophytes for sustenance (France 1996a). Problems with overlapping $\delta^{13}C$ values between terrestrial litter and attached algae due to boundary layer diffusion resistance in relation to water turbulence (France 1995b; Hecky and Hesslein 1995) occurs in other aquatic systems besides streams (France 1996b). Therefore, although $\delta^{13}C$ analysis has met with some success in identifying the incorporation of allochthonous forest detritus into biota residing in oligotrophic lakes wherein filamentous algae are rare (Rau 1980; France 1996c), especially if the $\delta^{15}N$ differences between potential food sources (France 1995c) are also taken into consideration (France 1996d), many of the problems that will be discussed here are not unique to stream research.

Study Comparisons and Interpretations

Characterization of the Representative Autochthonous (algal) $\delta^{13}C$ Value

Doucett et al. (1996) collected three samples of the filamentous algae *Cladophora* sp. in each of four study streams, which they then "presumed to be representative of the algal carbon signature at each site." *Cladophora* was used "because it was available in sufficient quantity at all sites and to avoid the previously reported difficulty of separating terrestrially-derived detritus from algae in rock scrapings."

There are four problems in this approach: a) because of the many factors that can affect algal $\delta^{13}C$, it is incorrect to expect that such a small sampling effort (n = 3) can capture the real isotopic variability at any single stream location (see Kline et al. 1990; Rosenfeld and Roff 1992); b) because algal $\delta^{13}C$ is at least partially regulated by water turbulence (France 1995b; France and Holmquist 1997), filamentous algae are often found to be depleted in $\delta^{13}C$ compared to other forms of periphyton (e.g., Hamilton and Lewis 1992; Mize 1993; France 1995d; France and Cattaneo 1998), and can never, therefore, be "representative" of any other algal $\delta^{13}C$; c) to not sample epilithon because of logistic difficulties ignores techniques that have been developed to separate such detrital and algal mixtures for isotopic analysis (Hamilton and Lewis 1992); and finally d) to assume that filamentous algae are even grazed in the first place is counter to common knowledge (France et al. 1991), and is the likely explanation as to why Doucett et al. found so many of their organisms to be ^{13}C-enriched compared to *Cladophora* (discussed further below).

Mize (1993), using a sampling effort for algae of up to 36 collections per study site, determined that a considerable variability existed in algal $\delta^{13}C$ with standard deviations being as great as 23% of their means. Because of this, Mize concluded that "autochthonous carbon is not always distinct from terrestrial carbon and autochthonous carbon is too complex to derive any meaningful single autochthonous value"; "taking one autochthonous component such as epilithic aufwuchs or algae that happens to be isotopically distinct from terrestrial carbon and using this one component as a basis for comparing differential utilization of allochthonous and autochthonous carbon forces the investigator into using faulty and incomplete data"; and "the present study points out the fallacies of trying to conjure single isotopic values for autochthonous carbon". Finally, Mize rhetorically questioned, therefore, "how can an investigator predict the isotopic values of any consumer that feeds exclusively on submerged aquatic vegetation when the food source itself has such a documented broad range of isotopic values?"

Relationships of Animal $\delta^{13}C$ to That of Their Food

Doucett et al. (1996) believed that the "higher $\delta^{13}C$ values in brook trout may have reflected their larger size and higher trophic position relative to other fishes at this site (Rau et al. 1983)", even after the *a priori* use of a trophic correction factor in their mixing model of +1 o/oo "between an animal and its food...(DeNiro and Epstein 1978)", where trophic position was "determined for each fish species and age class of salmon using the $\delta^{15}N$ data obtained in this study."

There are two problems in this approach: a) because marine organisms are ^{15}N-enriched compared to those occupying freshwaters (France 1994), "any particular $\delta^{15}N$ value in a fish is clearly not an inviolate marker of food-web position alone but is rather a reflection of a combination of both trophic and source fractionation" (France 1995e), especially in regions with anadromous fish such as the coastal streams studied by Doucett et al. (see Kline et al. 1990, 1993;

Hesslein et al. *in* France 1995e; Bilby et al. 1996); and b) although dietary ^{13}C-enrichment may occur in the laboratory (DeNiro and Epstein 1978) and be detected in the open ocean (Rau et al. 1983), it is erroneous to assume the inter-trophic position dynamics of δ^{13}C in open oceans are in any way transferable to situations in other systems (del Giorgio and France 1996), as demonstrated by the observation of weak or nonexistent trophic effects on carbon isotopic variability for coastal (Gearing et al. 1984; Dickson 1986; Wada et al. 1987; Fry 1988; Hobson and Welch 1992) and freshwater (France 1995f, 1996e) food webs (reviewed by France and Peters 1997).

Mize (1993) scrutinized the literature with much more rigor than did Doucett et al., stating: "DeNiro and Epstein (1978) concluded that the degree of accuracy in stable isotope technology is poor and that applications should be restricted to comparison of diet sources where large isotopic differences are found. Gearing et al. (1984) warned that the range of isotopic differences between consumers and food sources is too great to generalize with any certainty about how consumers fractionate stable carbon isotopes." This led Mize to conclude that "because there is no established, agreed-upon consumer-food source [carbon] isotopic fidelity paradigm, depending on which assumptions or theory one chooses, the same data can be interpreted in different ways."

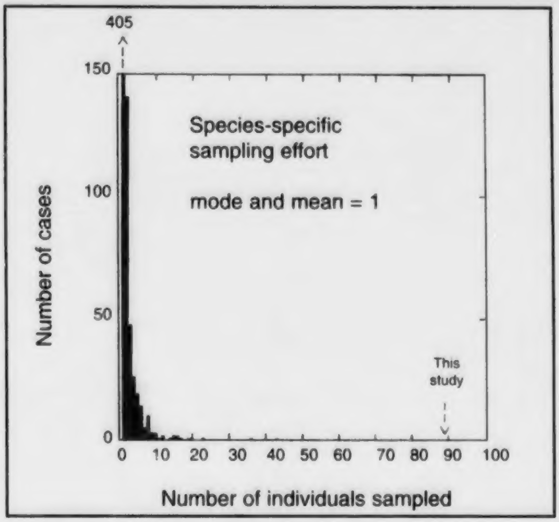

Figure 1. Sampling intensity used to characterize the carbon isotope ratio of freshwater species. Note that about half of these published δ^{13}C values are analytically unreplicated in terms of sub-samples. "This study" refers to crayfish from France (1996c).

Determination of Animal-Specific δ^{13}C Values

Doucett et al. (1996) based their site-specific ascriptions of organismal δ^{13}C on sample sizes of 6 to 20 for fish (across seasons) and unspecified numbers of pooled samples of up to 10 individuals for invertebrates.

Although the sample sizes used by Doucett et al. are generally typical for freshwater research (Fig. 1), such low numbers may give a false impression of species-specific isotopic fidelity, especially considering that pooling was used. More extensive sampling may have revealed δ^{13}C differences of over 10 o/oo among unpooled individuals from the same study-site as a result of either ontogenic diet shifts (France 1996c), variable foraging locations (France and Steedman 1996), or simply feeding idiosyncrasies (France 1996e).

Mize (1993), whom like Doucett et al., also used relatively small sample sizes to determine species δ^{13}C values, interpreted his findings, however, more cautiously: "because stable carbon isotopes do not move predictably through food webs (and because investigators have been led to believe they should), writers often make unsupportable speculations about feeding relationships"; "the claim that a consumers's δ^{13}C represents recent feeding history needs to be viewed with great caution. Given the uncertainty of carbon turnover time and the uncertainty and variability of isotopic adjustment to dietary changes, an investigator does not know what time frame of assimilated feeding history is reflected"; and "given the unknown tenure of 'isotopic memory', an investigator often cannot determine what carbon sources combined to produce the δ^{13}C value of any fish sample or tissue sample of other stream consumers."

Derivation of Isotopic Mixing Models to Determine the Differential Incorporation of Allochthonous and Autochthonous δ^{13}C

Doucett et al. (1996) determined "the relative contribution of terrestrial leaf litter inputs (percentage allochthonous) in the diets of fishes using a two-source isotopic mixing-model equation" for three of their streams "where leaf litter and *Cladophora* were shown to be isotopically distinct." Fish in the headwater reach were considered to be dependent on terrestrial inputs for 85–100% of their carbon base, whereas those from lower reaches were "less dependent on allochthonous carbon", thereby "confirm[ing] the increasing importance of autochthonous primary production at downstream sites, as would be predicted by the river continuum concept."

Notwithstanding the previous difficulties in Doucett et al.'s study as outlined above, there is a further problem with their mixing model. Namely, of the 18 samples of fish that were analyzed, only 4 of these actually had $\delta^{13}C$ values between the two putative endpoints in their mixing model. How one can accurately apply a mathematical mixing model when both putative sources lie to one side of the fish values under investigation?

Mixing models as advanced by Peterson and Fry (1987) are supposed to allow one to determine which of two possible source endpoints the intermediate $\delta^{13}C$ values of organisms lie closer to. Because autotrophs differ in their average stable nitrogen isotope ratios ($15_N/14_N$ expressed as $\delta^{15}N$ in relation to the atmospheric standard) $\delta^{15}N$ values (France 1995c), one can sometimes reconcile difficult questions concerning proportional ecotonal coupling with the simultaneous use of both $\delta^{13}C$ and $\delta^{15}N$ analyses in concert (France 1996e).

Mize (1993) was bothered by the theoretical and practical failings of two-source mixing models: "how can an investigator track carbon transfers in a food web when a single component of one category (algae of autochthonous) has a range of isotopic values that brackets both consumers and allochthonous carbon isotopic values?"; and "Peterson and Fry (1987) postulated that the stable carbon isotope technology requires that there be no more than two carbon sources available to a consumer; otherwise, the true carbon source cannot be determined...Faced with the many distinct carbon components...how does an investigator derive a single autochthonous carbon isotopic value? Most investigators select one dissolved inorganic carbon (DIC)-converting autochthonous carbon component that happens to be isotopically distinct from allochthonous carbon and use this single component to represent all autochthonous carbon in a stream...a grouping process [that] creates artificial isotopic values for both carbon categories, especially for autochthonous carbon." As a result, Mize dispensed with the technique entirely, concluding that "because of the variety of autochthonous production components and their different isotopic values, the proportional utilization of allochthonous and autochthonous resources could not be quantified" for his streams.

Final Conclusions about the Effectiveness of $\delta^{13}C$ Analysis

Doucett et al. (1996) optimistically concluded their study: "stable isotope analysis appears to have great potential as an effective fisheries management tool because it provides a means by which the energy basis for secondary production can be determined...Estimates from an isotopic mixing model equation showed that fishes obtained most of their carbon from allochthonous sources in the headwaters, but in downstream sites, autochthonous inputs played an equally important role. These results are consistent with the river continuum concept...As the [study site] is scheduled for logging in 1996–1999 ... continued sampling efforts should show the value of the stable isotope analysis technique for investigating trophic responses of fish communities to environmental change."

What Doucett et al.'s (1996) mixing model really describes is the increasing importance of marine carbon as one moves downstream; i.e., algae, invertebrates, and fishes in their streams all showed a progressive ^{13}C-enrichment with proximity to the ocean as would be expected. In retrospect, it is perhaps not surprising that Doucett et al. could not (in my opinion) definitively demonstrate the importance of allochthonous carbon given the extreme complexity of their coastal study region. For ecotonal coupling, it is difficult enough to attempt to understand what may be occurring when dealing with only two mixing sources such as either terrestrial and aquatic (e.g., France 1996c, d), or marine and freshwater (e.g., Kline et al. 1990; Bilby et al. 1996) environments. Attempting to understand ecotonal coupling resulting from the mixing of three environments, especially if one is relying solely on source differences in only a single isotope as in Doucett et al., is bound to be unsuccessful.

Mize (1993), despite working in a much less complex ecotonal milieu than that of Doucett et al., was extremely pessimistic (overly so in my opinion) in his interpretations of the wide-spread effectiveness $\delta^{13}C$ for determining allochthony: "this study and other similar studies use stable carbon isotopes to draw conclusions about carbon relationships in freshwater streams that do not stand up well under critical scrutiny…stable carbon isotope technology offers little or no meaningful improvement over traditional study methods such as gut content analysis, feeding observations, feces analysis, and radioactive tracers", such that "it is doubtful that stable carbon isotope technology, alone, can ever provide a valid, supportable analysis of autochthonous/allochthonous carbon resource utilization in freshwater streams at any meaningful level of resolution."

Scientific Psychology and Hypothesis Testing

Both of the studies detailed in this review are the product of first time graduate thesis contributions (Doucett 1994; Mize 1993). Science develops as the systematic presentation of immediate convictions determined through sense-perception (Popper 1968). This process operates best if approached phenomenologically with an objective assessment of data as accomplished by Mize, rather than fundamentally by a seeming need to find a positive result as by Doucett. This latter is part of the increasingly worrisome state of modern science wherein "there's no weather unless it's raining—there's no results unless they're positive." Results should never be forced to conform to a preconceived belief that science can only advance through the corroborative endorsement of positive findings.

Attempts to nonobjectively force data to concur with preconceived beliefs in any theory (i.e., confirmation bias) can occur even when disconfirming evidence that glaringly contradicts the hypothesis is clearly presented (Loehle 1987; DeMelo et al. 1992). Such a procedure, as in the case of Doucett et al. (1996), produces what Loehle (1988), borrowing from Kipling, refers to as "just-so-stories". Stable carbon isotope analysis, despite the blanket pessimism of Mize, may very well be able work in certain stream situations. Until we understand, however, more about the factors responsible for disagreements between theory and results, we must continue to treat attempts to discern allochthonoy in streams through $\delta^{13}C$ analysis with caution (Peterson et al. 1986; Winterbourn et al. 1986; Rosenfeld and Roff 1992; Mize 1993; France 1995a, 1996f).

Acknowledgments

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Hydrology and Temporal Distribution of Organic Carbon from an Upper Boreal Wetland Ecosystem

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Abstract

The Tri-Creeks Experimental Watershed is located in Townships 47 and 48, Ranges 22 and 23, West of the 5th Meridian, about 40 km southwest of Edson, Alberta. The Alberta Forest Service and related agencies of the Alberta and federal governments, in conjunction with a succession of Forest Management Area lease holders, conducted research into water and sediment yield changes and their effects on the salmonid fishery at the site from 1967 to 1986. This paper deals with dissolved and particulate carbon data collected over the last 3 years of the study, an extensive literature review on carbon regimes in both pristine and harvested watersheds, and one epoch of sampling conducted during October, 1995. Watershed management issues raised in the literature are discussed with respect to the Tri-Creeks data.

Headley, J.V., Sneddon, D.T., Neuwirth, M., and Korchinski, M. 1998. Hydrology and temporal distribution of organic carbon from an upper boreal wetland ecosystem. Pages 183–186 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 14, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

Research was conducted in the Tri-Creeks Experimental Watershed, Alberta from 1967 to 1986, to study water and sediment yield changes and their effects on salmonid populations. These studies were performed by the Alberta Forest Service and related agencies of the Alberta and federal governments, in conjunction with a succession of Forest Management Area lease holders. This paper discusses the results observed for trends in the dissolved and particulate carbon data collected over the last 3 years of the study. Watershed management issues are discussed with respect to the Tri-Creeks data.

In this work, investigations were focused on the assessment of the site-specific relationships between the hydrology, land-use (clear cutting), and the temporal distribution of organic carbon. Dissolved organic carbon (DOC) was considered to be an index parameter for assessing the assimilative capacity of natural streams and wetlands.

Study Area

The Tri-Creeks Experimental Watershed is located about 40 km southeast of Hinton, Alberta, at 53°9′ North latitude and 117°15′ West longitude, in the foothills of the Rocky Mountains (Fig. 1) (Jablonski 1978). It is about 60 km^2 in area.

The climate of the site is continental and strongly modified by topography. Annual precipitation averages 860 mm, 64% of which falls as rain during the summer months (Bodnaruk 1987). The snowpack generally begins to accumulate in mid-October, peaks in March at about 100 mm, and disappears in mid-May. An average of 60% of snowfall is lost to evaporation or melt before spring, a consequence of the strong chinook winds the area experiences.

The peak of freshet is generally finished by late April. The lowest flows appear to occur in February, although data on winter flows is scanty. A second low-flow period occurs in late August. While the contribution of groundwater to low flow was not established, isotopic data suggest virtually all baseflow is derived from shallow groundwater. Very little recharge of deep aquifers occurs in this area (Sterenberg et al. 1986). Water discharge is correlated strongly with geochemistry (Sneddon and Korchinski 1987) and the composition of the soils and bedrock in the watershed. Soils in the study area are moderately to very highly erodible luvisol and brunisols, commonly gleyed, and developed on weathered bedrock (Hudson et al. 1985). The dominant tree species is lodgepole pine (*Pinus contorta*),

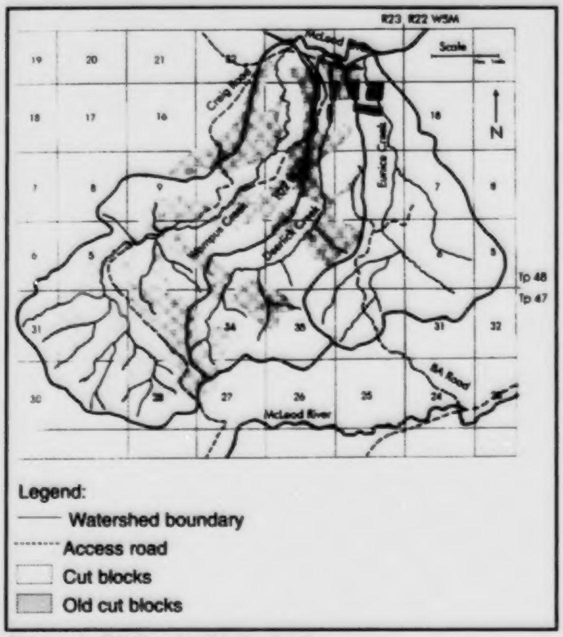

Figure 1. Tri-Creeks Watershed.

with occasional stands of subalpine fir (*Abies lasiocarpa*) in upland areas, and black spruce (*Picea mariana*) in the prominent wetland areas in the creek bottoms. Small stands of trembling aspen (*Populus tremuloides*) occur along the McLeod River at the outlets of the streams, and in isolated stands elsewhere in the study area. The study area extends through the Upper Boreal into the Subalpine forest zones. The Upper Boreal zone supports wetlands that cover approximately 30% of the study area.

Experimental Design

Experimental design was based on a paired watershed. Eunice Creek (16.1 km^2) served as the undisturbed control. Deerlick Creek (14.8 km^2) was treated by harvesting alternate clearcut blocks to the stream bank. Wampus Creek (28.2 km^2) was treated by harvesting alternate clearcut blocks leaving one chain wide buffer strips next to the stream. Logging practices used during the harvesting phase were those approved by the Alberta Forest Service for the McLeod Working Circle. They were identical to those used elsewhere by the Forest Management Area permit holder of the time, St. Regis Forest Products Ltd. of Hinton, Alberta.

For all sampling events, the flow of the streams was measured by the Water Survey of Canada (Nip and Hursey 1988). The levels of DOC were

determined using the NAQUADAT 06107 method (Alberta Environmental Centre 1985). Temporal profiles for the dependence of DOC and particulate carbon (PC) with time were estimated by multiplying monthly discharges (average flow multiplied by the time between sampling events) by the observed average values for the concentrations of PC and DOC, based on the procedure described by Hobbie and Likens (1973).

Results and Discussion

The temporal distributions of PC showed a small peak in June 1984, two large peaks in the summer of 1985, and another peak in the autumn of 1985 (associated with an early snowfall event). Particulate carbon discharged for Wampus Creek was 400% higher than that of Deerlick Creek, and 800% higher than Eunice Creek (the control basin) during the June 1985 peak. In the winter months, the quantity of PC discharged was very low, and similar for all three creeks. The corresponding levels of DOC in Deerlick Creek in July 1984 were higher than the other two creeks. During the remainder of the study, levels of DOC were highest for Wampus Creek, except for the winter months, during which levels were low and similar in all three creeks.

Particulate carbon in the Tri-Creeks was similar to PC found in other small headwater streams described in the literature. It appeared to be largely derived from leaf detritus, decomposing litter, and animal debris in the stream bed and banks. The overall ratio of PC/DOC was 6% for Eunice Creek and Wampus Creek, and 10% for Deerlick Creek. The corresponding value reported for Hubbard Brook watersheds was about 7% (Hobbie and Likens 1973). Particulate carbon was thus a minor component of carbon export from the study streams, most of the carbon loss occurring as DOC.

For both 1984 and 1985, peak carbon yield from each watershed occurred between August and September, the period of autumn leaf fall. In 1985 the pattern was complicated by storms. In that year, leaf fall peak occurred in August, before the main storm season and maximum flows in September and October. The carbon export rate per unit area from the control basin (Eunice Creek) was approximately constant at 1.38 $kg \cdot ha^{-1}$ throughout the observation period. In contrast, the rates from the treated sites were higher, at 3.88 $kg \cdot ha^{-1}$ for Deerlick Creek and 2.1 $kg \cdot ha^{-1}$ for Wampus Creek. These observations are similar to those reported for other disturbed montane basins (Hobbie and Likens 1973; Telang et al. 1981).

The most significant differences among the watersheds was the degree of surface disturbance. This surface disturbance leads to surface erosion and subsequently greater carbon export until the surface vegetation can re-invade the disturbed area (Hudson et al. 1985). While the upper reaches of Wampus Creek and Deerlick Creek are both pristine and oligotrophic, the lower reaches have been shown to support an active and diverse ecology (Dietz 1971; Sterling 1973). Significant increases in the mean size of rainbow trout were observed in cohorts entering the population following logging (Sterling 1990). This suggests increased biological activity that could in part explain the disappearance of excess DOC from the system.

Conclusions

Temporal variation in DOC and PC concentration and yield in the Tri-Creeks study area was generally similar to those observed for other montane basins in their natural state. Based on temporal variations observed in this study, environmental monitoring of seasonal and diurnal dependence of DOC and PC parameters can provide useful screening information on the sustainability of water quality in basins disturbed by logging.

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Impact of Timber Harvest on Sediment Deposition in Surface Waters in Northwest Montana over the Last 150 Years: A Paleolimnological Study

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Abstract

An analysis of lake sediment cores from three northwest Montana lakes reveals substantial correlation between the mass sedimentation rate (MSR) of fine sediments, and past human land disturbance activities. Accelerated logging was followed by increases in MSR of up to 14 times the pre-European settlement rate. While natural land disturbance activities (including floods and wildfires) appeared to have some impact on MSR, human activities (led by road construction) appeared to be the greatest contributor to increased MSR. Two of the study lakes (Whitefish Lake and Lake McDonald) showed MSR increases associated with land disturbance activities earlier this century, with some evidence of lessened impacts from more recent logging activities. The third study lake (Swan Lake), showed substantial increases within the last two decades. This is apparently the result of more recent logging activity.

Introduction

One of the biggest environmental concerns involving the timber industry is the potential for enhanced erosion and transport of sediment to streams and lakes. Numerous studies report increases in lake sedimentation rates which correlate with nearby land disturbance activities, such as timber harvest, plowing of fields, and road building (Davis 1975; Battarbee et al. 1983; Berglund 1986; Whitmore et al. 1994). Increased sediment loadings are considered undesirable as they may degrade gravel spawning habitat used by stream fish (Weaver and Fraley 1991; Anderson 1998). Furthermore, since sediments represent a major source of water-borne nutrients (Mortimer 1941; Perry and Stanford 1982), increased erosion and sediment transport may accelerate the eutrophication process in surface waters.

Current debate over the impact of timber harvest activity on water quality in northwest Montana and elsewhere is often hampered by a scarcity of quantitative data on conditions prior to logging. Without pre-logging water quality data, it is difficult to assess the impact of logging activity on specific lakes and streams.

Spencer, C.N., and Schelske, C.L. 1998. Impact of timber harvest on sediment deposition in surface waters in northwest Montana over the last 150 years: a paleolimnological study. Pages 187–201 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Many streams in the Flathead River watershed in northwest Montana contain sections which are flanked by steep, naturally occurring, unstable stream banks of sand, silt, and clay. The presence of these erosive deposits may result in naturally elevated sedimentation rates in the watershed. Evidence of this natural erosiveness comes from observing rivers in the area such as the middle fork of the Flathead River. This lies almost entirely in the Bob Marshall Wilderness Area, but may appear muddy and turbid during particularly high flows from spring run-off. Given these observations, some feel that sediment erosion associated with logging activities in the Flathead River watershed is likely minor in comparison to natural sedimentation rates. Others feel that timber harvest activities in the Flathead valley have likely caused significant increases in erosion and sediment transport, based upon research on logging activity in other regions (Anderson 1998). The present study was launched to help resolve this debate.

We used paleolimnological techniques to approach this research problem. This method was chosen because historical changes in surface erosion and sediment transport within the watershed of a lake have been shown to be reflected in changes in the sediment character and the rate of sediment accumulation in the lake (Berglund 1986). The vast majority of suspended stream sediments carried into large deep lakes are deposited within the quiescent lake environment. Thus, undisturbed sediments deposited on the bottom of lakes contain a record of the past history of sedimentation from their respective watersheds. Modern paleolimnological techniques allow the estimation of past sedimentation rates through detailed analysis of the record preserved in the lake sediments (Berglund 1986). A record of past changes is most evident in sediments deposited in the deep-water (profundal) region of the lake. This environment is much more stable than nearshore, shallow lake, or stream environments. Deep lake sediments may remain largely undisturbed for thousands of years. Another advantage of the paleolimnological approach is that the lake sediments can serve as integrators of cumulative impacts occurring not only over long periods of time, but also over an entire watershed (Hecky and Kilham 1973).

Other common research methods evaluating logging impacts frequently involve measuring water quality parameters at sampling sites on streams in the vicinity of logging operations. Results from these studies tend to be site specific, and may miss impacts, depending upon the frequency and duration of sample collection. For example, erosion and sediment transport are often highest during spring run-off or during large storm events. Stream sampling in the northern Rockies can be problematic at these times due to access difficulties caused by spring snowpack or the unpredictable nature of storm events.

By contrast, lake sediments contain an integrated record, and can serve as continuous monitors of cumulative impacts over an entire watershed.

Methods

Three lakes were chosen for study (Fig. 1), based upon the nature of human activity in their respective watersheds. Whitefish and Swan lakes contain forested watersheds in which human land disturbance has consisted primarily of logging and associated road building. Lake McDonald, located in Glacier National Park, was chosen as a control since its watershed has remained virtually free from logging activity.

Sediment cores were collected during the summer and fall of 1990 using a freeze coring technique modified from Shapiro (1958) and described in Spencer (1991). Upon retrieval, frozen cores were stored in coolers containing dry ice and transported to the lab. Several cores were collected from deep, offshore regions in each lake, well away from major tributaries. A single core from each lake was subsequently selected for detailed analysis. Criteria for core selection included evidence of minimal sediment disturbance, and appropriate length (>25–30cm). The selected cores appeared to be representative of whole-lake conditions in Whitefish and Lake McDonald. Distinct patterns of horizontal banding were visible in the same relative location in other cores collected from the same lakes (Spencer 1991). No banding was observed in cores collected from Swan Lake; therefore, visual comparisons between cores was not possible.

An electric band saw was used to cut each frozen core into a series of measured sections, approximately 1 cm thick. Core sections were placed in Whirl-Pak bags, thawed, and homogenized. Volumetric samples were removed from each core section with a graduated syringe, weighed to obtain wet density, and then freeze-dried to measure dry density.

The mean date of deposition was estimated for each freeze-dried section using ^{210}Pb dating techniques (Appleby and Oldfield 1983). Radioisotope activity was measured using low background gamma counting (Appleby et al. 1986; Schelske et al. 1994). Samples were counted with a well-type intrinsic germanium detector (EG&G Ortec) and a 4096-channel

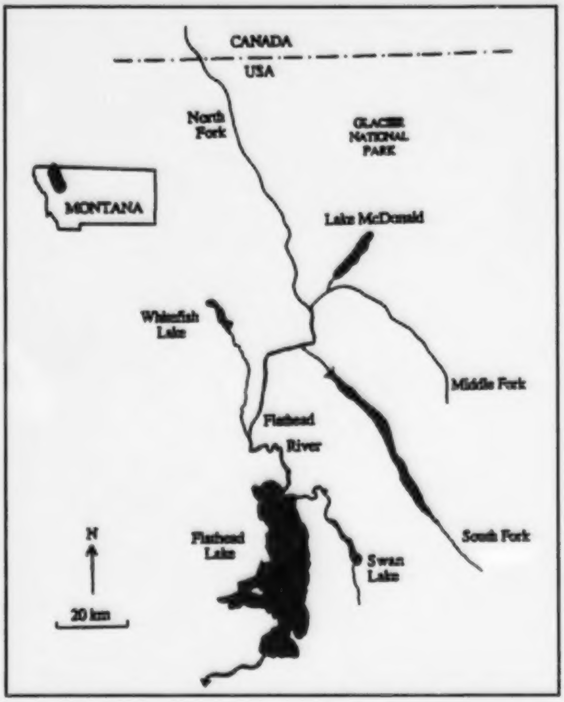

Figure 1. Map showing the state of Montana, USA, and an enlarged view of the northwestern part of the state showing the three study lakes (Whitefish, Swan and McDonald), located in the Flathead Lake watershed.

pulse height analyzer. Pre-weighed sediment samples were pressed into plastic test tubes to a nominal depth of 30 mm and then sealed with epoxy. Direct estimates of background (supported) ^{210}Pb activity were estimated using ^{214}Bi. Ages were calculated from the excess (unsupported) ^{210}Pb activity, using the constant rate of supply (CRS) model (Appleby and Oldfield 1983).

An independent technique was used for an alternate estimate of the location of sediments deposited in 1963. Atmospheric testing of atomic weapons peaked in 1963. Previous studies have documented a peak in ^{137}Cs activity in lake sediments deposited in that year, due to the global fallout of the radioactive decay particles (Pennington et al. 1973; Robbins and Edgington 1975). We found close agreement between the two dating methods (Spencer 1991; Schelske et al. 1994).

After the core sections were dated, mass sedimentation rates (mg dry wt cm^{-2} yr^{-1}) were estimated for the last 125–150 years. It is important to note that the present analyses were not designed to quantitatively establish whole-lake sediment budgets during the period of record. This would require analyses of a number of cores collected from various parts of each lake. Rather, the present analyses simply used the stable deep-lake sediments as a continuous monitor of relative changes in the erosion, transport, and deposition of fine sediment in the study lakes. For this reason, the units for MSR in figures 2,4, and 6 are given in relative units. However, it is possible to convert these to absolute units using data provided in the figure legends.

It is widely accepted that construction and maintenance of logging roads represents the largest source of increased sediment loadings from logging activity (Anderson 1998; Clarke et al. 1998b; Spillios and Rothwell 1998). Unfortunately, there are no quantitative records of logging road construction in the study watersheds. However, beginning with the mid-1900s, records were obtained containing the area of land subject to logging. An index of potential logging road impacts was developed using a 5-year amortization schedule developed by foresters at the Flathead National Forest. A direct correlation between timber harvest area and road construction has been reported by foresters in the Flathead Lake watershed, who developed sediment delivery coefficients for logging roads which decline over time as the roads stabilize (Mike Enk, Flathead National Forest, Bigfork, Montana, personal communication). Year one harvest area is assigned a value of 100%; year two, 60%; year three, 30%; years four and five, 15%; and years 6 and beyond, 0%. These coefficients were applied to timber harvest areas shown in figures 3 and 5 to derive a 5-year cumulative amortized index for timber harvest and related logging road impacts. Accurate harvest records are not available for early logging activity commencing around the turn of the century. For information on this early activity, interviews with old-time foresters were used.

Results

Whitefish Lake

The mass sedimentation rate (MSR) in Whitefish Lake has fluctuated widely over the last 140 years. The largest changes appear to be correlated with human land disturbance activities following European settlement, rather than natural disturbance events such as floods or wildfires (Fig. 2). The lowest MSR during the period of record occurred in the oldest dated core section (1850–1880). During this time period, the Whitefish Lake watershed was heavily forested, and human settlements in the watershed

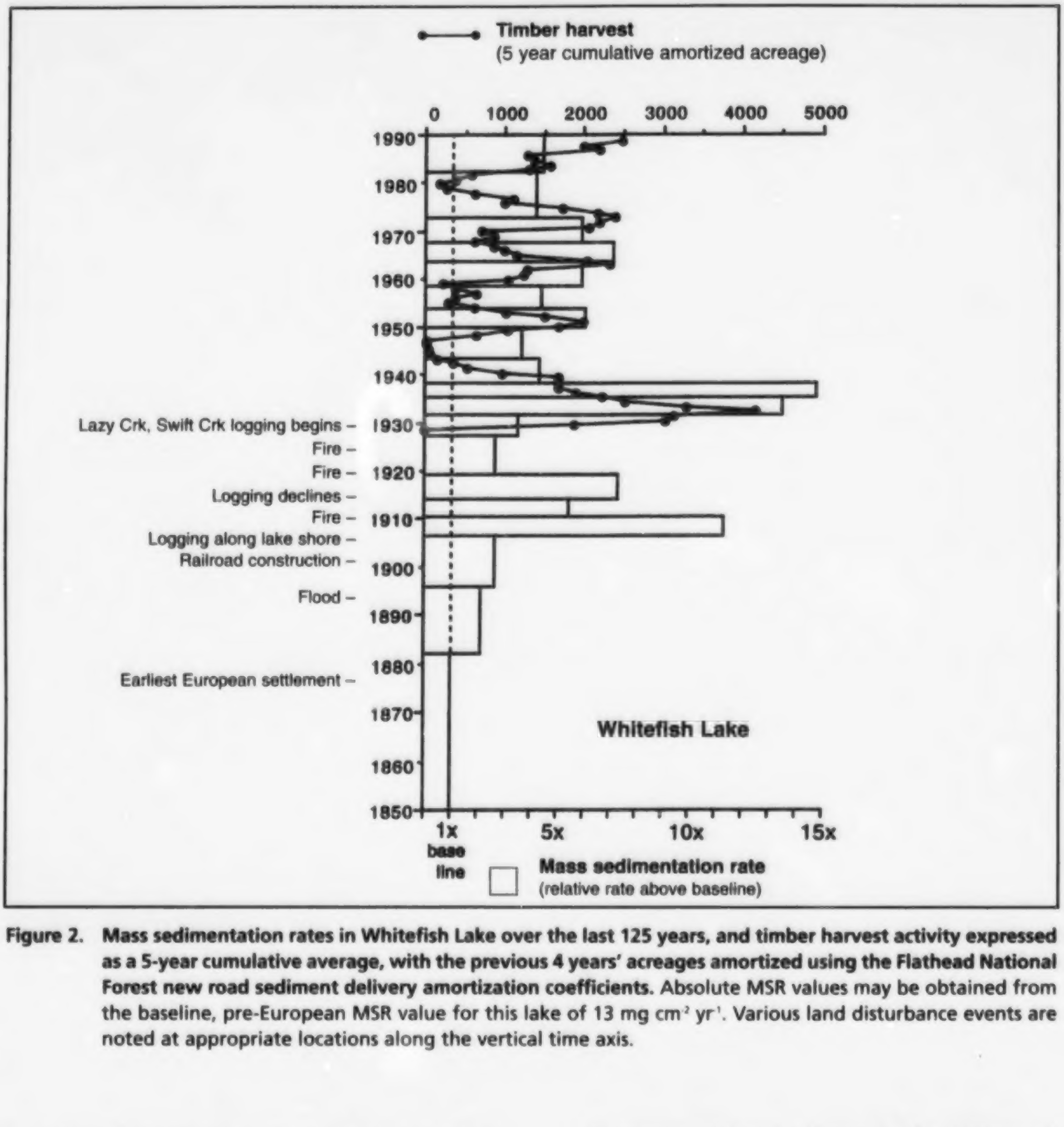

Figure 2. Mass sedimentation rates in Whitefish Lake over the last 125 years, and timber harvest activity expressed as a 5-year cumulative average, with the previous 4 years' acreages amortized using the Flathead National Forest new road sediment delivery amortization coefficients. Absolute MSR values may be obtained from the baseline, pre-European MSR value for this lake of 13 mg cm^{-2} yr^{-1}. Various land disturbance events are noted at appropriate locations along the vertical time axis.

were limited to an occasional small Native fishing encampment near the outlet of Whitefish Lake (Trippett 1956; Schafer and Engelter 1973). The largest flood during the period of our study occurred in 1894, when Whitefish Lake reached its highest recorded level, some 3 m above the low water mark (Schafer and Engelter 1973). There is evidence of a small increase in lake sedimentation during this flood. However, this increase was relatively minor compared to subsequent increases during the 1900s (Fig. 2).

European settlement began in the 1880s, and thereafter the MSR increased, remaining consistently above presettlement levels through to the present time. Peak sedimentation rates were recorded around 1910, and again in the 1930s, reaching levels more than 11- and 14-fold above baseline levels, respectively (Fig. 2). The sedimentation rate in recent times, though reduced from these earlier peaks, still remains 4–5 fold higher than pre-settlement levels.

1900–1929

Increased sediment delivery in the early 1900s is correlated with several significant land disturbance activities. In 1904, the Great Northern Railroad completed a rail line along the entire seven mile southern shoreline of Whitefish Lake. Construction activities along the steep southern lakeshore included the excavation of a tunnel near the head of the lake, the construction of a trestle, and the filling of Beaver Bay on the lake's southwest shore (Schafer and Engelter 1973). There is little doubt that substantial amounts of sediment were dumped or washed into Whitefish Lake during construction of the rail line.

In addition, logging activity around the lake during this time period also likely contributed to increased sediment loadings to the lake. Although quantitative records documenting the extent of this early logging activity could not be located, available information suggests that the activity was extensive in the immediate vicinity of Whitefish Lake. There were several sawmills in the area at this time, including one on Whitefish Lake near the outlet, one downstream on the Whitefish River, and several others farther downstream on the Stillwater and Flathead rivers. Logging activities in those days involved selective cutting of the larger trees near Whitefish Lake, which were pulled to the lake using horses (Schafer and Engelter 1973). The following two passages describe logging activities around Whitefish Lake in the early 1900s.

"Lumber that choked the lake in spring was carried across the ice in winter. In the spring of each year, big log drives took the logs down the Whitefish River.... Lumber companies were thick around Whitefish in those days and used to cut their logs on the lake then raft them down to the dam in great booms with steam and gasoline tug boats. Then when the logs reached the lower end of the lake they were held there till spring. When the ice went out in April or May and the water was running full, and the river was swift and unhindered, then they used to have log drives. They must have had them for at least ten years after we came to Whitefish before the timber was gradually thinned out and the mill at the dam took care of all that was left" (Schafer and Engelter 1973).

All of this activity likely contributed to increased sedimentation rates in the lake. The decline in lake sedimentation rate beginning around 1920 (Fig. 2) appears to correlate with the decline in logging activities in the watershed.

Wildfires may also have caused increased erosion in the watershed. Wildfire activity, however, has been relatively mild in the Whitefish Lake watershed, and does not appear to have had a large effect on lake sedimentation rates. Only three wildfires burned acreages greater than 1% of the lake watershed since the mid-1800s. The largest of these, affecting 6.7% of the watershed, occurred in 1910. The MSR did increase during the time interval of the 1910 fire (Fig. 2), and some of this increase may have been attributed to the fire. However, land disturbance activities (described previously) also occurred in the watershed at this time. Specific fire effects are therefore difficult to discern. The other two fires occurred in 1926 and 1919. They affected 3.7 and 3% of the watershed, respectively.

Although the 1926 fire was smaller than the 1910 fire, it was the only fire that produced an obvious layer of black ash, clearly visible in the sediment core (Spencer 1991). In addition to this thin layer of ash, local foresters and soil scientists indicate that the 1926 fire was the only one that left visible evidence of surface erosion, including several gullies that can still be seen today (Spencer 1991). In spite of these outward signs of erosion, the MSR during the time period of the 1926 fire was relatively low in Whitefish Lake (Fig. 2), again suggesting that human activities have had a much greater impact on MSR in Whitefish Lake than natural disturbances.

1929–present

Sedimentation rates in Whitefish Lake increased dramatically again in the 1930s (Fig. 2). This large increase corresponds with the extensive logging activities which commenced in 1929 in virgin timber stands in the Lazy Creek and Lower Swift Creek drainages, above the head of Whitefish Lake. Detailed logging records indicate that logging activity in the Whitefish Lake watershed was more extensive during this time period than at any other time (Fig. 2, 3). Stimuli for accelerated harvest at this time include improved transportation and a change in tax laws. The areas logged in the early 1930s were not logged previously, in part because the tributary streams to Whitefish Lake were relatively small and not suitable for carrying logs. Railroads and later trucks made it possible to log these areas. In the late 1920s, a railroad spur was constructed from the head of Whitefish Lake up the Lazy Creek drainage. Trees were cut, pulled to the rail line using horses, and transported to the mills by rail car. The Lazy Creek rail spur was operated for 3 years, and then removed

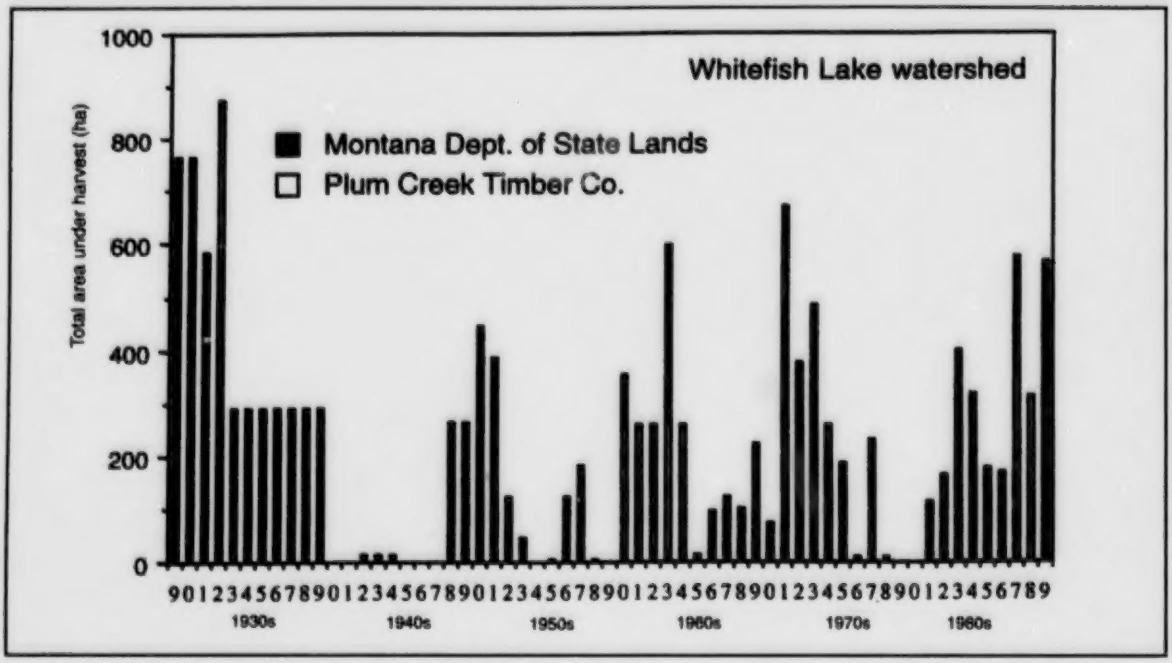

Figure 3. Timber harvest activity, by ownership, in the Whitefish Lake watershed during the period when accurate harvest records are available (since 1929). Plum Creek Timber Co. was formerly part of Burlington Northern Inc., which in turn was formerly part of Great Northern Railway Co.

in 1932 (D. Klehm, Whitefish, Montana, personal communication). Thereafter, logs were hauled out of the Lazy and Swift creek drainages by truck.

A change in property tax laws in the region may also have contributed to the accelerated timber harvest at this time. A new statute apparently placed a higher taxation rate on land supporting virgin timber than land in which trees had been removed. Following this change, timber harvest greatly accelerated in the Whitefish Lake watershed, primarily on private timber industry lands (Fig. 3).

During this time period, an extensive network of logging roads and skid trails were constructed above Whitefish Lake to remove the timber (Spencer 1991). At this time, there was little concern about water quality impacts, and few, if any, water quality regulations existed. Stream crossings were unrestricted, culverts were used infrequently, and "corduroy" roads were constructed across wet areas simply by clearing the trees and then placing timbers, side by side, across the wetlands to create a road bed (Spencer 1991).

In addition to a dramatic increase in MSR recorded in the mid-1930s, there was an obvious change in the character of the lake sediments at this time. This provides further evidence of large changes in erosional processes in the watershed. Corresponding with this time period, there was a very noticeable band of light gray sediments in the core, which had a higher density than the adjacent sediments. This band probably resulted from enhanced erosion of inorganic sediments (clay and silt) due to ground disturbance activities (Spencer 1991).

Thereafter, lake sedimentation rates declined in the late 1930s and early 1940s in concert with declines in timber harvest activity (Fig. 2). From the mid-1940s to 1990, timber harvest activities followed a cyclic pattern, with four distinct peaks in harvest activity in each of the succeeding decades (Fig. 2, 3). The first two peaks were in the early 1950s and 1960s. They were correlated with increases in sedimentation rates in Whitefish Lake, while the later peaks do not exhibit such a correlation. Logging activity in the 1950s and 1960s was concentrated in previously unlogged areas, and was accompanied by extensive road building. Increased surface erosion accompanying this activity likely contributed to the peaks in MSR in the early 1950s and mid-1960s (Fig. 2).

The 1964 flood occurred during the latter part of this trend. The sedimentation rate did not increase

dramatically following this flood, the second largest of the study period behind the 1894 event (Schafer and Engelter 1973). The sedimentation rate instead continued an increasing trend which was initiated in the 1950s. Although the 1964 flood may have contributed to increased erosion and sediment transport in the watershed, the impact of this flood on lake sedimentation appears small in comparison to previous land disturbance activities.

Unlike the earlier time periods, the peaks in timber harvest activity in the early 1970s and the late 1980s were not correlated with noticeable increases in lake sedimentation rate (Fig. 2). Lake sedimentation rates during this period were still 4–5 fold higher than pre-settlement levels. However, the absence of distinct peaks in sedimentation (noted previously) provides evidence that these more recent logging activities may have produced less surface erosion than previous timber harvest activities of similar magnitude. Several factors may have contributed to this change.

The 1970s and 1980s coincide with significant efforts on the part of government resource management agencies and the timber industry to reduce the impact of timber harvest activities on water quality. A combination of mandatory and voluntary standards were adopted to reduce the sedimentation risk. These efforts focused on minimizing erosion associated with road construction and stream crossings, and restricting harvest activities on the most sensitive lands. The data provide evidence that these measures may have reduced erosion and sediment transport to surface waters.

A second factor may have contributed to the reduced erosion. More recent logging activities concentrated on secondary timber growth on lands which had been previously logged, primarily in the 1930s. The result is that fewer new logging roads were built in these areas in the 1970s and 1980s, due to the old roadbeds from the previous logging activities. Thus, reduced construction of new roads may also have contributed to the reduced sedimentation rates in the 1970s and 1980s.

The overall impact of the accelerated logging activity in the late 1980s may not have fully manifested itself in the core, collected in 1990. Analysis of earlier activities revealed a lag of several years between the peak logging activity, and the peak MSR (Fig. 2). It is possible that sediments from this recent logging activity may still be in transit in the Whitefish Lake watershed. A subsequent large run-off event, such as one that occurred during the spring of 1995 (Fig. 7), could result in elevated MSR from earlier activities.

In addition to timber harvest activities, two other factors may have contributed to changes in lake sedimentation in recent years. First, the 1980 eruption of Mt. St. Helens produced a fallout of volcanic ash across western Montana. There is no visible ash layer in the study cores, and lake sedimentation rates did not appear to increase during this time period. Nevertheless, sediments were deposited in the watershed as a result of this eruption. Second, recent increases in lakeshore housing and other developments along Whitefish Lake may have contributed sediments to the lake. Although lake sedimentation rates did not increase dramatically during recent years, it is possible that observed sedimentation rates would have been lower in the absence of lakeshore development and volcanic ash deposition.

Swan Lake

As in Whitefish Lake, changes in mass sedimentation rates over the last 120 years in Swan Lake appear to be more closely linked with human activities than natural disturbance events in the watershed (Fig. 4). The highest MSR in Swan Lake occurred during the decades of the 1970s and 1980s, a time of greatly accelerated logging activity. Like Whitefish Lake, the lowest mass sedimentation rate during the period of record occurred in the oldest dated core section, corresponding to the late 1800s, prior to European settlement. This MSR was used as the baseline rate for the Swan Lake watershed in its undisturbed, heavily forested condition.

The Swan Lake watershed was more remote than the Whitefish Lake watershed. Europeans did not begin to settle the area until the early 1900s (A. Whitney, Bigfork, Montana, personal communication). Following European settlement, the MSR began a century-long period of increase (Fig. 4). The highest MSR occurred from 1972 to 1990, some threefold higher than the baseline, pre-settlement rate. This recent sedimentation increase is correlated with a large increase in logging activity and associated road building in the Swan Lake watershed (Fig. 5). During the 1980s, timber harvest activities doubled over the 1960s' and 1970s' levels. Timber harvest and associated road building expanded on bottom lands and on the steeper flanks of the Mission and Swan mountain ranges that enclose the Swan Valley. Much of this increased harvest occurred on previously unlogged, private lands (Fig. 5).

The trend of increasing MSR appears first during the 1920s, when sedimentation rates more than doubled over background rates. Possible contributing

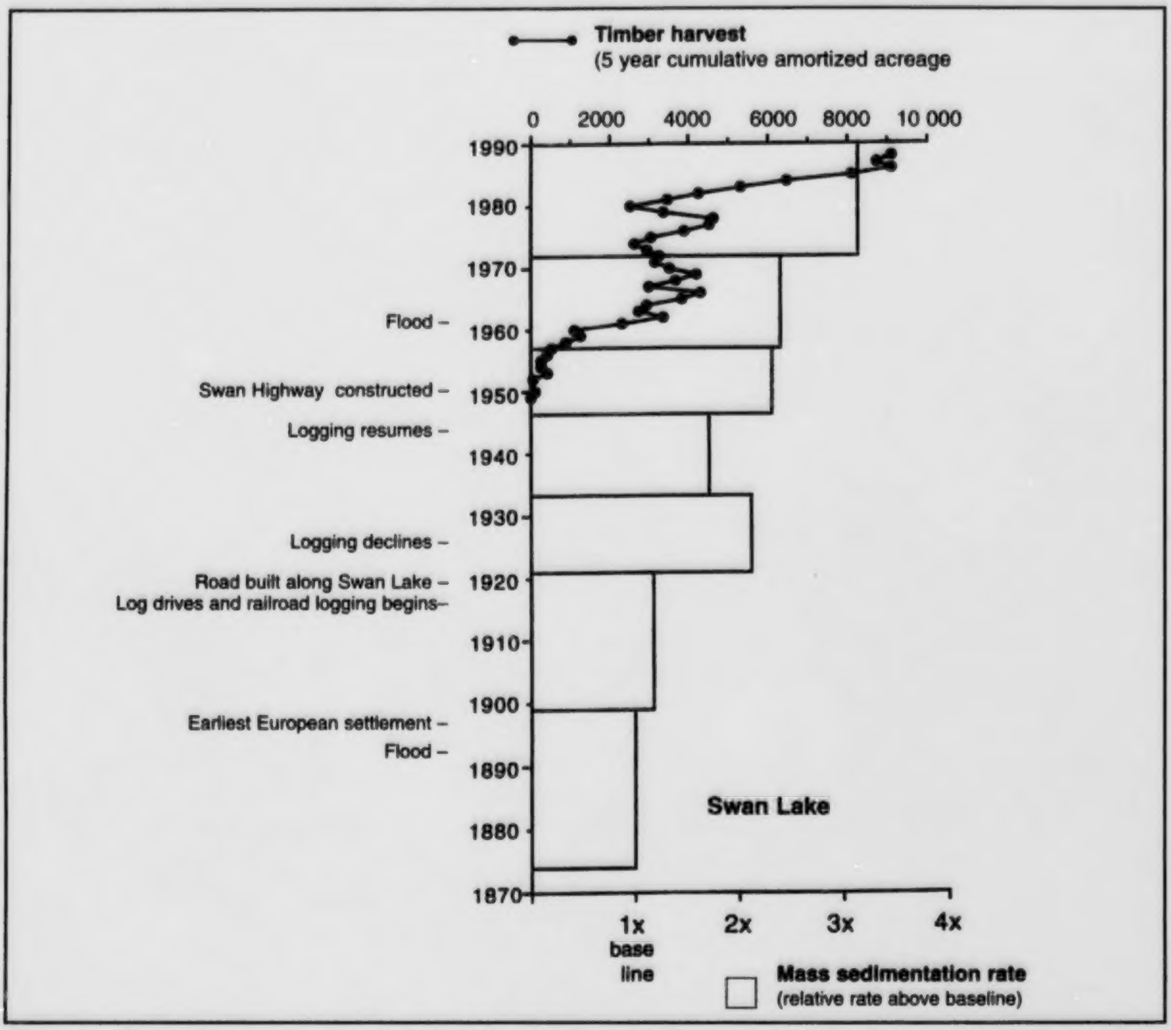

Figure 4. Mass sedimentation rates in Swan Lake over the last 125 years, and timber harvest activity expressed as a 5-year cumulative average, with the previous 4 years' acreages amortized using the Flathead National Forest new road sediment delivery amortization coefficients (as described in the methods section). Absolute MSR values may be obtained from the baseline, pre-European MSR value for this lake of 16 mg cm-2 yr-1. Various land disturbance events are noted at appropriate locations along the vertical time axis.

factors include: early logging activity in the watershed beginning around 1910 and continuing into the mid-1920s; and the construction of a primitive road along parts of the lake shortly before 1920 (A. Whitney, Bigfork, Montana, personal communication). During the late teens and early 1920s, logging activities intensified. A 3-mile long railroad spur was built from the head of Swan Lake up to South Lost Creek. Logs were cut and then pulled by horses down to South and North Lost creeks. In addition, a sluice dam was built on South Lost Creek during this period (A. Whitney, Bigfork, Montana, personal communication). Water was released from the dam during spring run-off and large numbers of logs were swept downstream into the Swan River and, subsequently, into Swan Lake. The increase in MSR recorded in the 1920s is consistent with early visual observations by a longtime local forester, who noted that the water in the Swan River near Bigfork used to run pretty clear even during spring run-off (A. Whitney, Bigfork, Montana, personal communication). The first time he recalled the river being murky was during the spring run-off of 1921 and 1922. Logging activity declined in the mid to late 1920s. It did not

resume again until the late 1940s, following construction of an improved highway into this remote watershed.

As in Whitefish Lake, natural disturbance events appear to have had much less impact on the MSR than human disturbance activities. The floods of 1894 and 1964 are not correlated with noticeable increases in MSR in this watershed. Fire activity in the Swan Lake watershed was even less pronounced than in the Whitefish Lake watershed. During our study period, there were no fires that burned more than 1% of the watershed, and there were no visible ash layers in the sediment cores.

Although timber harvest and related road building represent the largest human land disturbance activity in the watershed, there has also been an increase in the construction of lakeshore cabins and homes around Swan Lake, particularly in recent years. This activity may also have contributed to increased lake sedimentation.

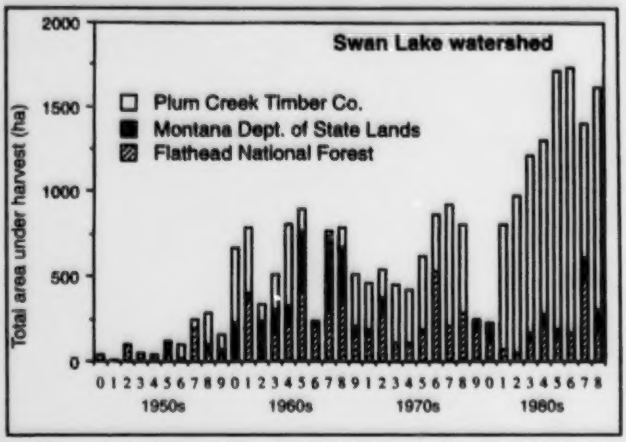

Figure 5. Timber harvest activity, by ownership, in the Swan Lake watershed during the period when accurate harvest records are available (since 1950).

Lake McDonald

Lake McDonald in Glacier National Park was initially selected as a control lake because there has never been any significant logging activity in the watershed. Unfortunately, another human disturbance in the watershed was overlooked: the construction of the Going to the Sun Highway. This highway appears to have been a major contributor to lake sedimentation in the middle part of this century (Fig. 6). As in the other study lakes, the lowest sedimentation rate in Lake McDonald was measured in the oldest dated section of the core, between 1880 and 1910. The MSR increased slightly between 1910 and 1935, and then increased substantially between 1935 and 1945, to levels more than four times the background rate. These increases appear to coincide with construction of the Going to the Sun Highway, which passes through the heart of the Lake McDonald watershed. This road runs along the south shoreline of Lake McDonald for much of its 9-mile length, and continues up the drainage along McDonald Creek for 11 miles, closely bordering the creek in numerous locations. The road then switches back up the long, steep, exposed terrain within the Lake McDonald watershed leading up to Logan Pass on the Continental Divide. At the time of construction in the early 1930s, this road was considered an engineering marvel, given the rugged mountainous character of the terrain it traverses. Roadbed preparation included the construction of numerous embankments along steep areas, blasting tunnels through bedrock, and considerable earth moving activities. All of this undoubtedly contributed sediments to surface waters in the watershed.

Although the initial road was completed in the early 1930s, the sedimentation rate in Lake McDonald did not peak until the late 1930s (Fig. 6). This apparent lag in lake sedimentation may have resulted from a delay in the transport of sediments from the upper portion of the watershed into the lake. Such delays may have been due to the long distance between Lake McDonald and the road building activities on the erosive slopes near the continental divide. In addition, the heavily forested streams in the McDonald Creek watershed may have higher sediment retention rates than those in logged watersheds, since natural downfall in the stream bed can serve as stream sediment traps. Numerous studies report reduced quantities of coarse woody debris in streams following logging activity (Clarke et al. 1998b). Furthermore, unlogged watersheds may have smaller maximum stream flows compared to logged watersheds (Swanson et al. 1998), which could result in reduced sediment flushing capacity in undeveloped watersheds. Thus, there are a number of factors which may have contributed to the delay in sediment transport, which appears to be more lengthy in Lake McDonald than the other study lakes.

The sedimentation rate in Lake McDonald declined rapidly in the 1960s. Revegetation and stabilization of the original road cuts likely reduced

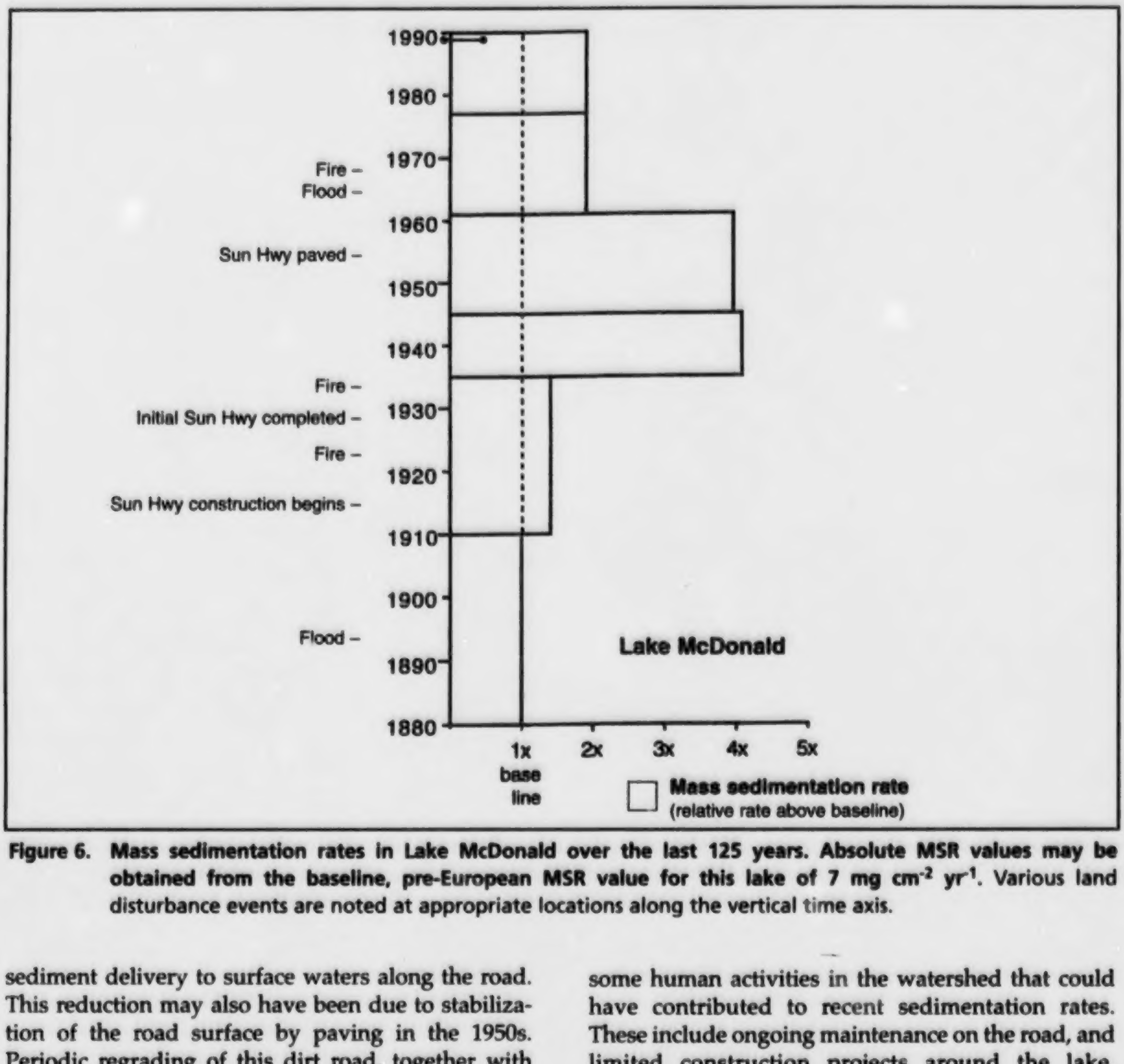

Figure 6. Mass sedimentation rates in Lake McDonald over the last 125 years. Absolute MSR values may be obtained from the baseline, pre-European MSR value for this lake of 7 mg cm^{-2} yr^{-1}. Various land disturbance events are noted at appropriate locations along the vertical time axis.

sediment delivery to surface waters along the road. This reduction may also have been due to stabilization of the road surface by paving in the 1950s. Periodic regrading of this dirt road, together with road dust stirred up by heavy traffic along this scenic roadway, may have contributed to elevated sedimentation levels in Lake McDonald. After the road was paved, the potential contribution of fine sediments to the lake would likely have been greatly reduced.

During the 29-year period from 1961 to the present, the mean sedimentation rate declined to slightly less than twice the background sedimentation rate (Fig. 6). Reasons for the continued existence of sedimentation levels above background are speculative. The 1980 eruption of Mt. St. Helens left no visible band of sediments in the core, though it is possible that this eruption caused increased sediment deposition in Lake McDonald. In addition, there have been some human activities in the watershed that could have contributed to recent sedimentation rates. These include ongoing maintenance on the road, and limited construction projects around the lake. Finally, it may simply take decades or more for the lake sedimentation rate to return to background levels following a major land disturbance like construction of the Going to the Sun Highway.

Natural disturbance events do not appear to be closely correlated with major changes in the sedimentation rate in Lake McDonald. As in the other study lakes, there were no large fires in the watershed during the period of record. One of the larger fires in the watershed occurred in 1967, when approximately 5% of the watershed burned (Barrett 1988). The 1964 flood also occurred during this interval. Sedimentation rates during the time interval following the flood and fire remained well below those

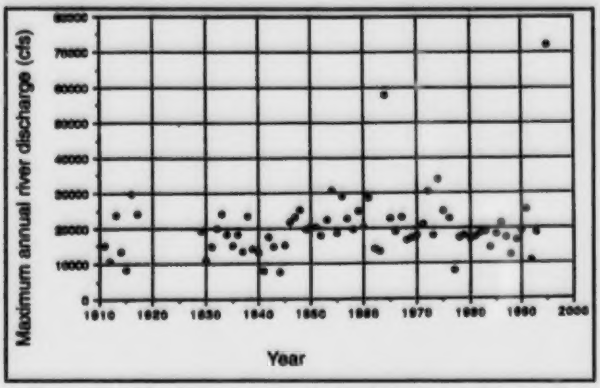

Figure 7. Maximum annual discharge in the north fork of the Flathead River at Columbia Falls from 1910 to 1995. The gauging station did not operate from 1918 to 1928. This gauging station has the longest stream flow record of any location in the study area.

achieved during the 1940s and 1950s. Smaller fires in 1926 and 1937 (Barrett 1988) may have contributed to increased sedimentation in Lake McDonald. However, the impact of these natural disturbances appear to have been masked by the road building activities.

None of the Lake McDonald watershed fires or floods left visible bands in the sediment core during the 110 year period of this study. However, a thick ash layer was visible in the Lake McDonald core at a depth of 18–19 cm. This is below the oldest strata dated in this study (Spencer 1991). Tree ring studies give evidence of several large fires in the Lake McDonald watershed in 1735, including one that burned much of the steep landscape leading down to the shoreline of Lake McDonald (Barrett 1988). The thick ash band likely corresponds to this fire. Given the appearance of this thick (1 cm) ash layer in the lake sediments, sedimentation rates must have increased dramatically in Lake McDonald following this large fire.

Discussion

The present study provides evidence for links between human disturbance activities (primarily related to timber harvest and road building) and increased fine sediment loadings in all three of the study lakes. Furthermore, these human disturbance activities appear to have had a much greater impact on fine sediment deposition in lakes than natural disturbance events such as floods and wildfires. Similar conclusions have been drawn from paleolimnological studies in other regions (Hutchinson et al. 1970; Davis 1975; Battarbee et al. 1983; Berglund 1986).

The lowest sedimentation rates in all of the lakes occurred in the 1800s, prior to European settlement. Thereafter, sedimentation rates increased at various times due to different land disturbance activities. During some time periods, the correlation between logging activity and deposition of fine sediments in the study lakes is striking. This correlation leaves little doubt as to a cause and effect relationship between the two. For example, the highest sedimentation rates recorded in Whitefish and Swan lakes occurred near the time of maximum timber harvest activity in the 1930s, and 1970s and 1980s, respectively.

The construction of new roads is widely cited as one of the greatest sources of sediment erosion associated with timber harvest. The pioneering paleolimnological study by Hutchinson et al. (1970) documented increased sedimentation rates in Lago di Monterosi in Italy that coincided with construction of a Roman Road (the Via Cassia) in 171 B.C. Results from the present study also show evidence of this link between road building and erosion. Construction and use of the heavily traveled Going to the Sun Highway, through the heart of the rugged Lake McDonald watershed in Glacier National Park, are correlated with peak lake sedimentation rates in the 1930s until the 1950s, when the road was finally paved. Deposition of fine sediment in Whitefish Lake appears to be closely correlated with the harvest of virgin timber stands, and attendant logging road construction, especially from the 1930s through the early 1960s. Using this 30-year trend of cause and effect, the sedimentation deposition in Whitefish Lake did not rise as much as expected during subsequent timber harvest in the 1970s and 1980s. However, timber harvest at this time was concentrated on secondary growth in areas previously logged in the 1930s. The reduced sedimentation may have been due in part to the fact that new road construction (a major source of sediments) was reduced because of pre-existing roads in the drainage.

Other changes in the 1970s and 1980s may also have helped lessen the impact of the more recent logging activities on sediment transport to Whitefish Lake. This time period coincides with the enactment of a series of mandatory and voluntary water quality and timber management practices designed to reduce erosion. The data from Whitefish Lake provide some evidence for reduced sedimentation rates from newer forest practices. The data from Swan Lake, however, are less convincing since the highest

MSR occurred during the last two decades of harvest activity (Fig. 4). Thus, if recent forest practices employed in the 1970s and 1980s did significantly reduce sediment delivery, then any such improvement appears to have been offset by the recent large expansion of harvest activities into new areas of the Swan Lake watershed.

Additional cautions are in order regarding the effectiveness of newer logging practices in reducing sediment transport. These include the fact that recent sedimentation rates are still well above the background sedimentation rates prior to European settlement. This suggests that the impacts of historical land disturbance activities appear to be cumulative, and may continue to be manifested for long periods of time. Peak sedimentation rates in our study lakes were sometimes delayed beyond the 5-year period used in the current amortization schedule for assessing sediment loadings from new roads. This was especially noticeable in Lake McDonald. There, sediment apparently originating from a roadway completed in the early 1930s, continued to be delivered to the lake into the 1950s (Fig. 6).

There is a second caution concerning the evaluation of recent timber management practices. It relates to the fact that the late 1970s through to the 1980s were characterized by a series of relatively mild runoff years (Fig. 7). It is possible that sediments resulting from harvest activities of the 1970s and 1980s may still be in transit, temporarily stored in the watershed. If this is the case, then data from our cores (collected in 1990) may underestimate the full impact of the more recent timber harvest activities on sediment deposition in Whitefish and Swan lakes. This hypothesis should be evaluated by collecting new cores from the study lakes, especially in light of a large flood which occurred during the spring of 1995 (Fig. 7).

Although timber harvest (and related activities) is the primary land disturbance activity in the Swan and Whitefish watersheds this century, both watersheds have also experienced other human activities that could have produced increased sediment deposition. The most notable activity, especially in recent years, is the development of lakeshore cabins and home sites along both Whitefish and Swan lakes. Comparisons between these lakes yield additional insight into the potential impact of lakeshore development on lake sedimentation rates. It is possible that the relatively high sedimentation rate recorded in Swan Lake over the last two decades may be due in part to increased lakeshore development. However, Whitefish Lake has experienced similar lakeshore development in recent years, yet recent sedimentation rates do not appear to have increased as they have in Swan Lake (Fig. 2, 4). It is unlikely, therefore, that lakeshore development is responsible for the recent large increase in sedimentation in Swan Lake. Rather, the contrasting lake sedimentation responses appear more closely related to differences in the history and timing of timber harvest and road building activities in the two watersheds.

Ecological and Timber Management Implications

Prior to this research, there was considerable debate in northwest Montana concerning the relative impact of human activities versus natural processes on sediment loadings to surface waters. Our data clearly indicate that the impacts of human activities are quite visible above the background noise of natural disturbance events. Over the last century, the impact of natural disturbance events are barely discernible in the sediment record of the study lakes. In contrast, human activities are correlated with pronounced changes in sedimentation rates. Our results support the conclusions of other studies that identify road building as the largest potential source of sedimentation associated with timber harvest (Anderson 1998; Spillios and Rothwell 1998; Clarke et al. 1998b).

The increased transport and deposition of fine sediments to surface waters may cause a variety of adverse ecological impacts. First, the increased deposition of fine sediments in fish spawning areas may substantially reduce the spawning success of stream fishes. Weaver and Fraley (1991) demonstrated that the emergence of bull trout (*Salvelinus confluentus*) and westslope cutthroat trout (*Salmo clarki lewisi*) fry from spawning gravels was significantly reduced by the addition of fine sediment. These species are native to the Flathead Lake watershed. The bull trout was recently listed as a threatened species under the Federal *Endangered Species Act* while the west slope cutthroat trout is designated as a species of special concern by the state of Montana. Furthermore, Weaver and Fraley (1991) report that timber harvest activities have had a quantifiable, negative impact on streambed composition and salmonid populations in northwest Montana. Their conclusions are supported by the paleolimnological data from the present study. This data documents the increased deposition of fine sediments in surface waters following road building, timber harvest, and other land disturbance activities throughout this century.

Another ecological concern produced by increased sediment loadings to surface waters is the potential for the undesirable stimulation of algal productivity and lake eutrophication. Sediments are a

major source of nutrients to surface waters (Mortimer 1941; Perry and Stanford 1982). Data from the Flathead Lake watershed show a close correlation between suspended sediment concentrations in streams and stream nutrient (i.e., phosphorus and nitrogen) concentrations (Spencer and Hauer 1991; Ellis and Stanford 1988a; Stanford and Ellis 1988; Stewart 1983; Golnar 1985). Bioassay experiments demonstrate that the addition of sediments from a variety of locations in the Flathead Lake watershed stimulate algal growth in lakes (Perry and Stanford 1982; Ellis and Stanford 1988a, b). Thus, enhanced erosion and sediment transport, as documented in this study, have almost certainly contributed to lake eutrophication. This is an undesirable process that can lead to reduced water clarity, oxygen depletion, and other water quality problems (Wetzel 1988).

Two of our study lakes show early symptoms of lake eutrophication. This likely resulted, in part, from past increases in sediment loading. Dissolved oxygen measurements in Swan Lake show severe hypolimnetic oxygen depletion in the south basin of the lake, with dissolved oxygen levels measured as low as 0.5 mg L^{-1} near the bottom of the south basin in 1990 (Spencer 1991). This is the lowest dissolved oxygen concentration that the senior author has measured across a number of large oligotrophic lakes in the northwest Montana, noted for their high water quality. There are no historic records of oxygen measurements in Swan Lake, and therefore it is not clear when the oxygen levels there first became depleted. Nevertheless, the reduction of hypolimnetic oxygen levels to near anaerobic conditions in the lake is surprising and alarming. It led to Swan Lake being reclassified as an impaired lake by the Montana Water Quality Bureau.

Further increases in inputs of sediment and nutrients to Swan Lake from timber harvest, road building, or other sources could easily exacerbate the dissolved oxygen depletion problem. If hypolimnetic oxygen levels were to decline just a bit more, and the conditions become anaerobic at the sediment-water interface, a rapid deterioration of water quality in the lake could be expected. Chemical changes associated with anaerobic conditions would cause the release of large quantities of phosphorus from the lake sediments (Mortimer 1941; Wetzel 1988), furthering the lake eutrophication process. This negative scenario has been documented in numerous lakes in other parts of the country, frequently fueled by increased human activities in lake watersheds (Vollenweider 1968; Horne and Goldman 1994).

At present, Whitefish Lake is in a transitional state between oligotrophy and mesotrophy (Golnar 1985). Late-summer hypolimnetic oxygen depletion is already occurring in the lake, but not to the extent of Swan Lake. Golnar (1985) concluded that Whitefish Lake lies near a critical threshold of "excessive" phosphorus loading, as determined from the nutrient loading model of Vollenweider and Kerekes (1980).

Mass sedimentation rates remain 3–4 fold above background rates in both the Whitefish and Swan lake watersheds. Given current ecological concerns surrounding increased sediment loadings to surface waters, the results show the need to immediately reduce sediment loadings to surface waters in these watersheds. This should involve careful planning and close scrutiny of any future timber harvest activities, and other land development activities that are becoming increasingly evident in the region. Given the link between road construction and increased sediment loadings, the few remaining roadless areas should be left roadless, thereby eliminating the single largest source of erosion resulting from timber harvest activities.

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The Takla Fishery/Forestry Interaction Project: Study Area and Project Design

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Abstract

While coastal-based fisheries/forestry interaction research has been conducted since the early 1960s, the relationship between forest harvesting and the productive capabilities of aquatic habitats in the interior of B.C. are poorly understood. In order to assist with the development of interior fish, forestry, and wildlife guidelines, and to test the efficacy of B.C.'s new Forest Practices Code, a new research project was initiated in 1990 on five tributaries of the Stuart–Takla watershed. The watershed basins are in the Hogem Range of the Omineca Mountains at the northern end of the Sub-boreal Spruce biogeoclimatic zone (Engelmann Spruce–Subalpine Fir zone at high elevations). The Stuart–Takla watershed supports both early and late run sockeye salmon, a distinct race of kokanee, and many other species of salmonids (e.g., rainbow and bull trout) and non-salmonids (burbot, squawfish, and shiners). This is a long-term multidisciplinary project that is spatially (five creeks) and temporally (before and after a variety of forestry treatments) controlled. Forestry activities will begin in two of the watersheds during the winter of 1996. Project components are designed to develop an understanding of the ecosystem processes that effect stream production and forest outputs. Participants from a number of agencies are involved including the B.C. Ministry of Forests, Fisheries and Oceans Canada, Carrier Sekani Tribal Council, B.C. Ministry of Environment, Lands and Parks, and several of B.C.'s universities.

Macdonald, J.S., and Herunter, H.E. 1998. The Takla Fishery/Forestry Interaction Project: study area and project design. Pages 203–207 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

Although research on the interaction between coastal-based fisheries and forestry has been conducted since the early 1960s, the relationship between forest harvesting and the productive capabilities of aquatic habitats in British Columbia's (B.C.) interior are poorly understood. To assist with the development of fish, forestry, and wildlife guidelines for the interior, a new research project was initiated in 1990 on four tributaries of the Stuart–Takla watershed (Figs. 1, 2). These tributaries were Bivouac, Gluskie, Forfar, and O'Ne-ell creeks. In recent years, with the introduction of the B.C. Forest Practices Code (FPC), the project has evolved to consider a wider geographical area: Baptiste and Van Decar creeks (Fig. 2), and a greater variety of streams types. A major research goal is to test the effectiveness of the recent FPC legislation in protecting aquatic resources. Of particular interest are small tributary systems (classified as S4 and S6 streams in the FPC, Table 1) that receive little riparian protection under the FPC prescriptions.

Following the proven pattern of previous coastal research (Hartman and Scrivener 1990), the Takla Fishery/Forestry Interaction Project (TFFIP) incorporates long-term (7+ years), integrated, multi-disciplinary approaches. Participants from a number of agencies are involved, including: the B.C. Ministry of Forests; Fisheries and Oceans Canada; the Carrier Sekani Tribal Council; the B.C. Ministry of Environment, Lands and Parks (MoELP); Canadian Forest Products Ltd. (CanFor); and several of B.C.'s universities. These partnerships have been pivotal in the success of the project to date.

Physical Description

The study area is in the Hogem Range of the Omineca Mountains, at the northern end of Sub-boreal Spruce biogeoclimatic zone. The growing season is short compared to coastal zones in B.C. The experimental watersheds are largely undisturbed. No logging has occurred in the watersheds, with the exceptions of Bivouac and Gluskie creeks (which had partial logging in 1993 to control an insect outbreak), and Van Decar (where limited harvesting occurred during the 1980s) (Mr. D. Roy, CanFor Ltd., Fort St. James, B.C., personal communication). A forest fire swept through portions of study area in the late 1800s (Mr. D. Roy, CanFor Ltd., Fort St. James, B.C., personal communication).

The Stuart–Takla system forms the northern most drainage of the Fraser River basin. The areas of the experimental watersheds range from 36 to 75 km^2, and average creek length is about 20 km. The lower reaches are designated under the FPC as S2 (greater than 5 and less than 20 m in width), and are generally about 8 m wide. Average daily water temperature in the creeks ranges from 0 to 13°C, with summer maximums near 18°C.

Aquatic Resources

The Stuart–Takla watershed supports both early and late run sockeye salmon (*Oncorhynchus nerka*), a distinct race of kokanee (*O. nerka*) (Foote et al. 1989), and many other species of salmonids [e.g., rainbow (*O. mykiss*) and bull trout (*Salvelinus confluentus*)], and non-salmonids [burbot (*Lota lota*), squawfish (*Ptychocheilus oregonensis*), and minnows (*Cyprinidae*)] (Macdonald et al. 1992). Early Stuart

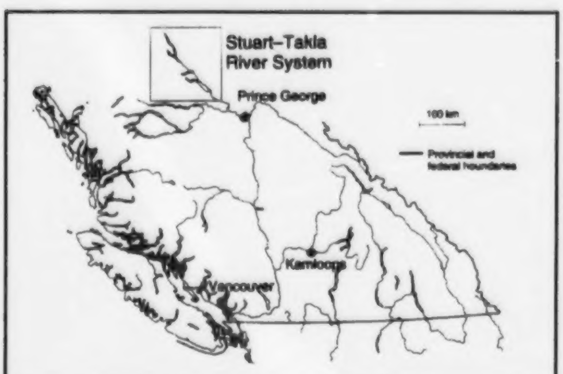

Figure 1. Fraser River watershed showing the location of salmon streams, the major cities, and the Stuart–Takla watershed.

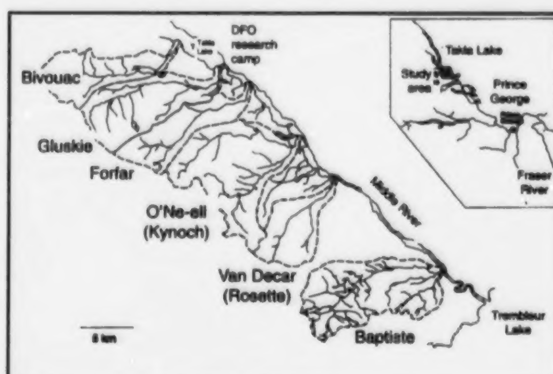

Figure 2. Takla Fishery/Forestry Interaction Project study area showing the experimental streams and watersheds.

Table 1. The following chart lists the research partners and projects by subject. Projects are grouped within the watershed processes that they share in common, to demonstrate research linkages. The researchers are aware of each other's activities and have maximized the opportunities for collaboration and synergy.

Project	Project leader	Agency[a]	Stream type[b]
Physical			
Suspended sediment			
Riparian treatment impact on suspended sediment regimes	Beaudry	MoF	S2, S4, S6
Sediment delivery from roads and cutbanks	Heinonen	DFO	–
Soil disturbance in different types of riparian management	Poulin	DFO	S2, S4, S6
Upslope sediment delivery processes	Hogan	MoF	Unknown
Stream sediment budgets	Beaudry	MoF	S2
Multi-element geochemical analysis of forestry effects	Fletcher	UBC	S4, S6
Settling dynamics of fine grained particles	Petticrew	UNBC	S2
Sockeye incubation gravel composition	Herunter	DFO	S2
Incubation gravel composition impacts behind beaver dams	Herunter	DFO	S2
Bedload			
Riparian treatment impact on bedload regimes	Beaudry	MoF	S4, S6
Upslope bedload sources	Hogan	MoF	Unknown
Bedload composition and volume	Herunter	DFO	S2
Bedload transport dynamics	Gottesfeld	UNBC	S2
Magnetic tracer rock monitoring techniques	Gottesfeld	UNBC	S2
Channel morphology surveys	Hogan	MoF	S2
Fluvial history on forested floodplains	Gottesfeld	UNBC	S2
Solar radiation			
Riparian treatment effects on stream temperature	Macdonald	DFO	S4, S6
Mainstream temperature monitoring	MacIsaac	DFO	S2
Riparian treatment effects on groundwater temperature	Macdonald	DFO	–
Watershed budgets in harvest/control creeks (Gates Creek)	Macdonald	DFO	S2, S4, S6
Hydrology			
Riparian treatment effects on groundwater storage	Macdonald	DFO	–
Rate-of-cut impacts on stream flow	Heinonen	DFO	S4, S6
Riparian treatments effects on stream flow	Beaudry	MoF	S4, S6
Annual water budgets in experimental watersheds	MacIsaac	DFO	S2
Floodplain coring to re-create past hydrological events	Petticrew	UNBC	S2
Snow survey monitoring	Thompson	MoELP	–
Snow accumulation; forest vs. cutblock	Heinonen	DFO	–
Water chemistry fluctuations	MacIsaac	DFO	S2
Temperature, permeability, and D.O.[c] in incubation environments	Macdonald	DFO	S2
Anchor ice depth/locations as impacted by harvesting	Macdonald	DFO	S2
Climatology (air temperature, radiation, precipitation)	Heinonen	DFO	–
Large and small organic debris			
Upslope vs. riparian origin	Hogan	MoF	S2
Stream reach and watershed surveys/aerial photos	Hogan	MoF	S2
Organic decay processes in riparian zones	Petticrew	UNBC	S2
Mechanization to increase wind firmness in riparian management zones	Poulin	DFO	–

Table 1 continued

Biological measurements			
Salmonids			
Sockeye stock enumeration; adults	Shubert	DFO	S2
Sockeye stock enumeration; fry outmigration	Whitehouse	DFO	S2
Sockeye egg development rates and success	Macdonald	DFO	S2
Sockeye alevin inter-gravel behavior	Herunter	DFO	S2
Fry survival in beaver impacted stream reaches	Herunter	DFO	S2
Fry habitat use, outmigration behavior and diets	Herunter	DFO	S2
Modeling temperature impacts on sockeye fry development	Macdonald	DFO	S2
Sockeye spawner distribution, abundance, and habitat requirement	Tschaplinski	MoF	S2
Redd depth, characteristics, and location	Herunter	DFO	S2
Sockeye spawner redd site selection	Macdonald	DFO	S2
Energetic capacities and limits of mature sockeye	Farrell	SFU	S2
Sockeye energy source analysis using stable isotopes	Johnston	MoELP	S2
Forestry induced stress and distribution of resident salmonids	Mellina	UBC	S4, S6
Integration of Takla findings to Fraser fish model	Williams	DFO	S2, S4, S6
Prey items			
Invertebrate population cycles and biodiversity	Macdonald	DFO	S2
Floodplain habitat evaluation; benthic/emergent insects	Fuchs	UBC	S2
Takla Lake zooplankton analysis	Shortreed	DFO	-
Stream periphyton and nutrient analysis	MacIsaac	DFO	S2, S4, S6
Riparian biodiversity			
Avian utilization of the riparian zone	Wiebe	CWS	S2

[a] MoF = B.C. Ministry of Forests; DFO = Fisheries and Oceans Canada; UBC = University of British Columbia; UNBC = University of Northern British Columbia; SFU = Simon Fraser University; MoELP = B.C. Ministry of Environment, Lands and Parks; CWS = Canadian Wildlife Service.

[b] S2 = A fish bearing stream that is greater than 5 and less than 20 m in width. The lower portions of the Takla experimental streams are typically S2 streams. S4 = A fish bearing stream less than 1.5 m in width. Some of the upper tributaries of the experimental streams are designated as S4 streams. S6 = A non-fish bearing stream less than 3 m in width. Some of the upper tributaries of the experimental streams are designated as S4 or S6 streams.

[c] D.O. = dissolved oxygen.

sockeye salmon enter the Fraser River in late June to late July, and spawn in early August after migrating almost 1200 km. Escapements of early Stuart sockeye salmon frequently exceed 10 000 fish to Gluskie, Forfar, O'Ne-ell, and Van Decar creeks while Bivouac and Baptiste creeks are less productive (P.A. Harder and Associates 1989). Peak escapements to Gluskie and O'Ne-ell creeks have exceeded 50 000 fish (Macdonald 1992). Sockeye salmon spawn in the lower 2 to 4 km of the study creeks. They are spatially separated from the late Stuart sockeye salmon, which spawn in the Middle River. Of the 30 tributaries used by the early Stuart sockeye salmon run, the TFFIP experimental creeks can support up to 50% of the total run (Tschaplinski 1996). Fertilized eggs hatch in October to November, and alevins overwinter in the gravel environment. In May, fry emerge and migrate downstream. They reside in lakes for 1 year prior to smoltification and migration to the ocean (Macdonald 1994).

Study Design

During the first 5 years of the project, prior to forest harvesting, the natural fluctuations and variability of many biological and physical parameters were monitored. Forestry activities are planned to commence in late 1996 in two treatment watersheds (Gluskie and Baptiste creeks) (Fig. 2). Forfar Creek will remain unharvested as a reference watershed for the life of the project. The experimental design therefore includes both a spatial and temporal control.

To assess the impacts of forestry on stream ecosystems, an incremental harvesting approach has been adopted. Initially, logging will begin adjacent to small (S4) systems, to study the effects on these small

tributaries and their groundwater supplies. In later years, increased logging will occur in the larger (S2) systems. All harvesting will follow FPC prescriptions. The initial harvesting will occur during winter months at the onset of freeze up, and will continue until spring thaw.

Study Components/Linkages

The research project components are designed to develop an understanding of the ecosystem processes that effect stream production and forest outputs. The components span a variety of physical and biological disciplines and, in the interest of research synergy, are designed to be complementary (Table 1). This process-oriented approach, examines the influence of natural actions and anthropogenic activities on physical parameters, and the eventual biological outcome. Biological assessments include measurements of sockeye salmon spawning, incubation, and rearing habitats, and the success and habitat use of egg, alevin, and fry life history stages. Physical components of the study include suspended sediment, bedload, solar radiation, hydrology, and organic debris. Each of these physical factors can be linked with an eventual biological outcome or effect. For example, physical factors such as suspended sediment, bedload, and organic debris may be inherently linked to channel morphology, which in turn may have an effect on the quality and quantity of fish habitat and fish production. Factors such as solar radiation and hydrology may have direct effects on system water regimes, which may impact spawning, egg and alevin survival, fry outmigration, and invertebrate production.

Summary

This paper serves as an introduction to nine papers in this session that will describe some components of the Takla Fishery/Forestry Interaction Project. The papers are presented in order, from physical processes to biological effects. They demonstrate the project's process-oriented approach and linkages. There are many other study components, in addition to these presentations (Table 1). The early stages of the project design have been described in detail by Macdonald et al. (1992). The published proceedings of two workshops provide further insight into a variety of the project components (Bernard et al. 1994; Macdonald 1994). For more information on the TFFIP, including a project bibliography, please contact the authors.

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Forest Management and Channel Morphology in Small Coastal Watersheds: Results from Carnation Creek and the Queen Charlotte Islands

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Abstract

Synoptic channel surveys on the Queen Charlotte Islands and intensive pre- and post-treatment studies at Carnation Creek on Vancouver Island show there is a direct, distinct, and permanent link between landslide occurrence and channel morphology. Landslides deliver large amounts of sediment and debris to these coastal streams, and this initiates the formation of large woody debris jams. Specific morphological changes occur upstream and downstream of the jams, impacting fish spawning, egg incubation, and rearing habitats. Logging on steep hillslopes accelerates landslide frequency and this has led to a corresponding increase in the number of recently formed debris jams. Streams associated with these young jams are characterized by extensive riffles, shallow pools, less stable bars, and an increased frequency, extent and duration of dry channel beds. All these characteristics contribute to fish habitat degradation. The influence of debris jams on channel morphology changes over time as jams deteriorate. Channel morphology is radically altered during the first decade following landslide inputs. However, morphology begins to resemble undisturbed conditions after approximately 35 years. Complex and diverse channels are typical after 50 years. The complexity is a result of the jam deterioration. Management of coastal watersheds has historically led to a shift in the age distribution of debris jams. Future management must ensure that any shift in landslide frequency, and therefore the debris jam age distribution, be minimized to maintain channel and fish habitat integrity.

Hogan, D.L., and Bird, S.A. 1998. Forest Management and Channel Morphology in Small Coastal Watersheds: results from Carnation Creek and the Queen Charlotte Islands. Pages 209–226 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

There is a direct link between stream channel morphology and in-stream fish habitats. Pacific salmon, trout, and char (salmonids) use stream environments for specific phases of their life cycle. Special conditions are needed for successful spawning, development and hatching of eggs, and growth and survival of their young (Toews and Brownlee 1981). Salmonids spawn in riffles composed of clean, stable gravels with well-oxygenated streamflows. Certain species also require stable pools to rear in for periods of time ranging from months to years. Young fish use pools to hide from predators, to feed in, and to generally grow in before migrating to the sea. Adult fish require an unobstructed migration path between the ocean and the stream spawning grounds. Similarly, young salmonids (fry and juveniles) require access along the stream channel, and into tributaries and sidechannels.

The central goals of forest and watershed management in British Columbia have been: to minimize changes in sediment and debris production and delivery to streams; to avoid changes to runoff patterns; and to eliminate the direct disturbance of channel banks and beds. To accomplish this, harvesting plans are now designed to: avoid landslide and erosion prone terrain; limit harvest rates; ensure high standards of road building and maintenance; and prohibit tree falling and yarding adjacent to streams (British Columbia Government 1995). Past practices, however, were not as carefully applied, and serious environmental impacts on stream channels and aquatic ecosystems have been widespread (Tripp 1994).

Most previous studies of the response and recovery of small streams to past forest management have concentrated on identifying specific channel impacts and their logging-related cause. They have generally speculated at the temporal duration of the impact. For example, studies in the western United States and Canada show that in most cases the channel bed aggrades, bank stability is reduced as the channel widens, and pools are infilled due to increased sediment inputs from landslides (Sullivan et al. 1987). The time required for these streams to recover to predisturbance conditions varied from five to over 60 years. The recovery time depends upon input sediment characteristics (e.g., location along the stream system, amount and particle size distribution) and the form and structure of the riparian zone.

In addition to changing sedimentation and hydrological characteristics of a watershed, logging has been shown to have a pronounced influence on the introduction and storage of large woody debris (LWD) to streams. The influence of LWD pieces on channel morphology has been investigated for more than two decades (cf. Thomson 1991). Large woody debris is an integral component of stream ecology (Hartman and Scrivener 1990). It influences both the physical and biological characteristics of small and moderate sized streams (i.e., channels with widths less than or equal to the length of in-stream debris). In comparison, little has been written about the importance of LWD accumulations (log jams) on channel morphology.

This paper reviews studies conducted in small coastal streams on the Queen Charlotte and Vancouver Islands. The Queen Charlotte Islands studies were designed to identify and describe the long-term response of stream channels to increased sediment loadings from both natural factors and forest management practices. The streams have a range of natural disturbance histories, with hillslope failure events documented back to the 1820s, and impacts from logging that began 40 years ago. The main focus is on LWD jam characteristics (i.e., origin, function, and longevity). These characteristics are a major factor controlling the long-term evolution of channel morphology and fish habitats.

The Carnation Creek Fish/Forestry Interactions Program channel studies were established to document annual changes in a series of study areas within a single watershed. The physical setting of Carnation Creek is similar to those in the Queen Charlotte Islands. The annual channel surveys enable much finer temporal resolution of channel change than the studies on the Queen Charlotte Islands. The Carnation Creek results are used to provide detailed, annual results, compared to the longer term (decades) results from the Queen Charlotte Islands.

Coastal Study Watersheds

The Queen Charlotte Islands are located approximately 80 km west of Prince Rupert in north coastal British Columbia (Fig. 1). The islands are characterized by abundant salmon-producing streams (Northcote et al. 1984), large areas of steep terrain underlain by highly erodible bedrock (Alley and Thomson 1978), and several soil types that are prone to mass movement (Wilford and Schwab 1982). The incidence of slope failure is also high, because the islands have a predominately wet climate, and experience frequent seismic activity. The average annual precipitation exceeds 3600 mm along the west coast, but Williams (1968) estimates this may reach

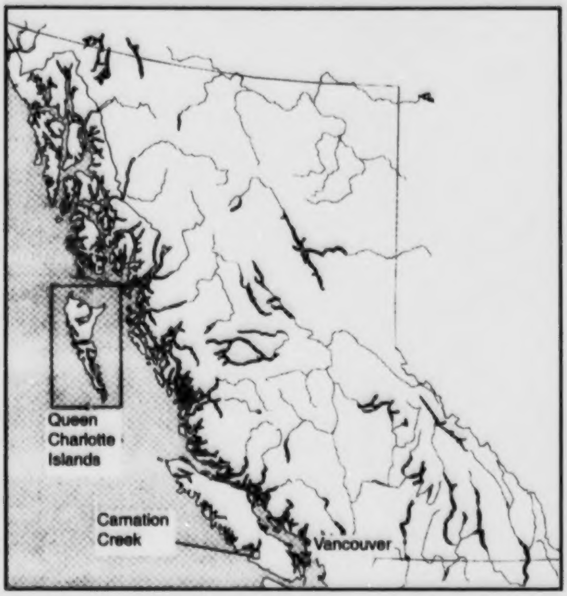

Figure 1. Location map showing the Queen Charlotte Islands and Carnation Creek Fish/Forestry Interaction Program sites.

7000 mm on coastal mountain ranges. The seismic activity is due to the location of the Queen Charlotte faultline separating the Juan de Fuca/Explorer and America plates (Sutherland Brown 1968; Church 1998).

Extensive surveys began in 1988 to explore the temporal and spatial relations between stream channel morphology and forestry activities. The results are given in Hogan (1989) and Hogan et al. (1995, 1998a). Channel surveys were undertaken over longitudinal profile lengths of 2740 channel widths (Wb) (a total of 43.7 km) in 12 watersheds on the Queen Charlotte Islands (Fig. 1). The surveys included channel lengths of 1193 Wb and 1547 Wb in forested and logged watersheds, respectively. Generally, watersheds were selected for analysis by their range of biophysical and land use characteristics (Table 1). Four of the watersheds are intact old-growth forests (one of which has a very small logged area away from all streams). The remainder have experienced various levels and methods of intensive logging over the last 50 years. The older logging methods involved large clear-cuts and high lead yarding systems, and usually included cutting to the stream bank. Buffers along the stream banks were not common in British Columbia prior to 1988.

Carnation Creek (Fig. 1) drains into Barkley Sound on the west side of Vancouver Island. A complete description of the Carnation Creek study is included in Hartman and Scrivener (1990). The Carnation Creek watershed is in the Coastal Western Hemlock (CWH) biogeoclimatic zone. It is small and has no lakes. The local climate is per-humid, and 95% of the annual precipitation falls as rain. Monthly stream flows are highly variable, ranging from 0.025 m^3s^{-1} in summer to 33 m^3s^{-1} during winter freshet. Peak flows up to 64 m^3s^{-1} have been measured. The basin is characterized by irregular topography with a wide, flat valley downstream, confined channels in the mid-valley, and steep valley walls with bluffs and rock outcrops in headwater areas. The bedrock is primarily volcanic with thin, coarse-textured soils that are well drained in most non-alluvial locations. Landslides are prevalent in the headwater and canyon areas.

Detailed longitudinal profiles were surveyed with an automatic level, and stadia rod. Distances were measured with a surveyor's hip chain. Several measurements were made to quantify channel morphology at set intervals along the channel (details included in Hogan 1989). The survey interval of one bankfull channel width was selected objectively from regional drainage basin area-channel width relations for forested and non-mass wasted watersheds (Hogan 1986). Water surface, bar top, and bank top elevations were determined at each one bankfull channel width interval. The b-axis of the largest surface stone visible on the bed was also determined.

All LWD, including jams, steps and individual pieces, were categorized at each survey interval according to their: size and shape (length, mean diameter and condition of root wad); position in the channel (orientation and vertical position); and function (bed and bank sediment trapping or scouring). A LWD jam is defined as a major accumulation of debris (either currently or over the last decades; remnants are still evident) that alters (or has altered) channel morphology and downstream sediment transport. The age, volume, and location of each jam was also recorded. Jam age was determined from the ages of nurse trees and bar/bank vegetation. A standard increment tree bore was used to obtain cores at, or near, the point of germination of the tree.

All morphological features (e.g., pools, glides, riffles, and runs) were surveyed along the longitudinal profile. Morphological breaks were included in addition to the set interval survey point. These were identified in the field by their topographic, sedimentological, and hydraulic characteristics, as defined by

Table 1. General characteristics of selected Queen Charlotte Islands watersheds

Creek	Basin Area (km^2)	% Steep Land	% Valley Flat	Relative Relief (m)	Precipitation (mm/year)	Physiography	Geology: Dominant	Geology: Sub-dominant	% Basin logged	Years since logging: started	Years since logging: stopped	Logging method	Logging to channel bank
Vancouver Island													
Carnation	11.2	39	6	800	>2100	ECP	Bo	–[a]	45	1976	1981	Cable yarding	Yes
Queen Charlotte Islands													
Government	16.5	24	15	490	>3600	QCR	KBI+SC/KU		0	–	–	–	–
Gregory	35.0	24	2	850	>3600	SP	Y	S+M	2	15	9	High leads	Yes
Inskip	10.3	70	1	1050	>3600	QCR	KBI+SC/KU		0	–	–	–	–
Jason	9.3	57	6	840	>3600	QCR	KBI+SC/KU		0	–	–	–	–
MacMillan	6.0	25	0	655	>3600	SP	H	HO	77	46	8	High leads	Yes
Mosquito	17.6	55	15	1140	>3600	QCR	K	–	65	45	25	Skid & High leads	Yes
Peel	11.2	71	1	1050	>3600	QCR	K	M+BI	15	30–40	30–40	High leads	Yes
Riley	27.6	29	12	870	>3600	SP	Y	CS+M	12	14	9	High leads	Yes
Schomar	7.3	25	4	655	>3600	QCR	Y	–	36	26	15	High leads	Yes
Shelly	5.2	56	0	750	>3600	SP	Y	S+CS	17	19	18	High leads	Yes
SBD	3.7	16	0	655	>3600	SP	HO	H	82	30	10	High leads	Yes
Tarundl	10.9	18	0	990	>3600	SP	C	BI+SC	37	27	9	High leads	Yes

[a]Dashes indicate not available.

Notes:

Physiography: SP – Skidegate Plateau; QCR – Queen Charlotte Ranges; ECP – Estevan Coastal Plain
For more details see Sutherland Brown (1968) and Holland (1976).

Geology: HO – Honna (conglomerate, sandstone)
H – Hadia (clastic sedimentaries)
S – Sandilands (orgillite, siltstone, tuff, sandstone)
Y – Yakoun (shale, siltstone, sandstone, conglomerate, volcanics);
K – Karmunsten (mafic volcanic flows, flow breccia, pillow flows, limestone)
C – Cretaceous shale (shale, siltstone, sandstone)
BI – Burnaby Island (diorite, monzodiorite, equigranular, quartz)
SC – San Christoval Plytonic (diorite, monzodiorite, equigranular, quartz)
KU – Kunga Group (argillite, siltstone, sandstone, tuff)
M – Masset (volcanics, aphyric, mafic to felsic flows, pyroclastic)
CS – Cretaceous Sandstone (sandstone, conglomerate, shale)
Bo – Bonanza (Jurrasic volcanics)

Keller and Melhorn (1973) and Sullivan (1986). Bankfull, wetted and valley floor widths were measured at the five bankfull channel width interval. General changes in riparian vegetation size and species were also noted.

The Connection Between Hillslopes and Stream Channels

Forestry Activities and Landslides

Forestry activities can influence the amount, timing, and nature of sediment and water moving through a stream system. As a result, channel morphology can be altered. This in turn can lead to habitat degradation. The impacts of forestry activities have been studied intensively over the last several decades (e.g., Salo and Cundy 1987; Hartman and Scrivener 1990; Hogan et al. 1998b). In general, logging and related activities have led to increased levels of sediment entering channels. Excess loads of coarse textured materials tend to promote bed aggradation. This leads to expanded bars and riffles, infilled pools and bank erosion. The gravel composition of riffles can become less suitable for egg incubation, due to increased proportions of fine sediments (<1 mm) within the gravels. Egg to fry survival rates can also be reduced because the enlarged riffles are less stable, and more prone to deep scouring down to the level of egg deposition. Logging debris left along streams can block mainstem and sidechannel access.

The Queen Charlotte Islands (QCI) have vast tracts of valuable commercial timber which, for over half a century, has made logging an important economic resource. The Coastal Western Hemlock biogeoclimatic forests consist of western hemlock, Sitka spruce, amabilis fir, and western redcedar, each of which are harvested extensively. Unfortunately, the inherent instability of the steep Queen Charlotte Islands hillslopes has been increased locally by logging (Schwab 1983; Gimbarzevsky 1986; Rood 1984). Rood (1984) emphasizes the relative importance of mass wasting as the dominant geomorphic process in steep areas, and documents a 34-fold increase in the frequency of mass wasting occurrences in logged areas. His results also indicate that there is 43 times more sediment (derived from hillslopes) entering stream channels in logged areas than forested locations.

Mass wasting events on the Queen Charlotte Islands occur episodically. There is ample evidence of historical landslides in the island landscape (Fig. 2). Gimbarzevsky (1986) inventoried almost 9000 landslides from a series of aerial photographs (the first set of photographs was from 1939) on the Queen Charlotte Islands. Schwab (1998) sampled 970 of these landslides and determined their date of occurrence by dendrochronological field surveys. His results (Fig. 3A) show that almost 85% of the total volume of sediment and debris derived from the landslides and delivered to stream channels was generated in seven large events occurring throughout the last two centuries (1810 to 1991). Of these, the largest events (in decreasing order of magnitude) occurred in 1917, 1891, 1875, 1978, and 1935; only the 1978 event post-dates the onset of logging.

The landslides documented by Schwab (1998) occurred during years that experienced severe rainstorms. The combined federal Atmospheric Environment Service records (Fig. 3B), beginning in 1887, show large storms in 1891, 1917, 1935, 1952, and 1978. Septer and Schwab (1995) provide a complete history of each storm. The 1891 storm lasted for three days, and produced 305 mm of rainfall in the first 24 hours. There were five major multiple-day storms in 1935, and at least three in 1917 along the north coast. The October 29-November 1, 1978 storm caused an estimated 1000 landslides on the Queen Charlotte Islands alone (Schwab 1983), due mainly to logging practices (particularly road building and harvesting on steep, gullied terrain) and very short duration rainfalls (120 mm/12 h.).

Landslides and LWD jam formation

Large woody debris jams are formed in coastal streams by several mechanisms. The most important factors are related to watershed attributes, particularly the link between hillslopes and stream channels. Channelized debris flows (torrents) introduce large amounts of sediment and debris to the channel (Fig. 2B). In watersheds with limited valley flat extent and hillslopes coupled directly to the stream system, jams are formed primarily at the terminus of debris flows that enter the channel. A significant relation between the terminus of a debris flow and the occurrence of a LWD jam has been identified in the Queen Charlotte Islands (Hogan et al. 1995). In coupled streams, much of the volume of debris in the jam is derived from upslope and upstream. Relatively minor amounts are derived from the proximal riparian area.

In decoupled watersheds where debris flows from steep hillslopes do not reach the stream, LWD jams originate from floated debris that become anchored at some point along the channel.

Figure 2. Examples of landslides and debris delivery to stream channels. A) Photograph of an old landslide track on the Queen Charlotte Islands. B) Photograph of sediment and debris delivered directly into the stream (coupled hillslope and stream). Large woody debris jam formed at the terminus of the debris flow.

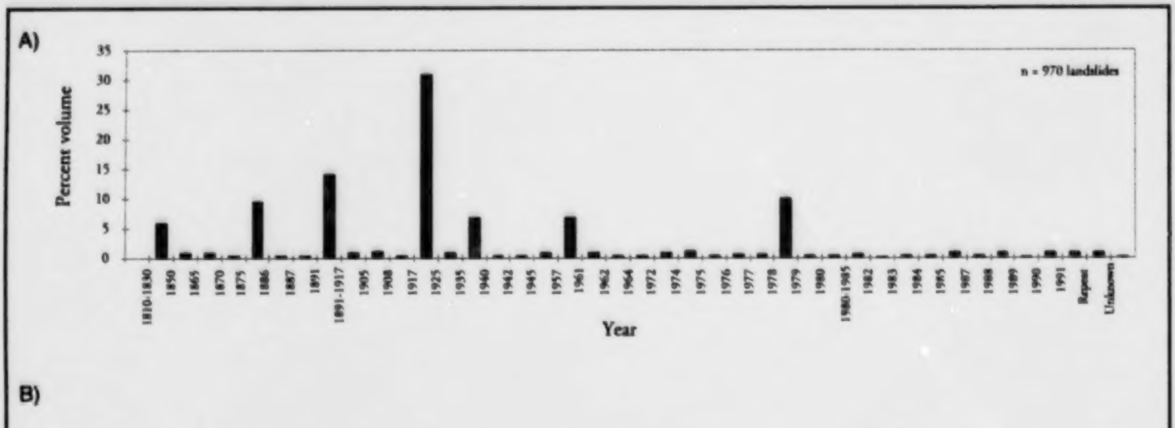

Figure 3. Historical landslide and precipitation records. A) Landslide events occurring in the Queen Charlotte Islands, 1810–1991 (from Schwab, 1994). B) Annual maximum 24-hour precipitation records for selected stations (aggregate record: Port Simpson, 1887–1909; Masset, 1910–1914; Queen Charlotte City, 1915–1948; Sandspit, 1949–1962; Tasu, 1963–1972; Sewell Inlet, 1973–1989).

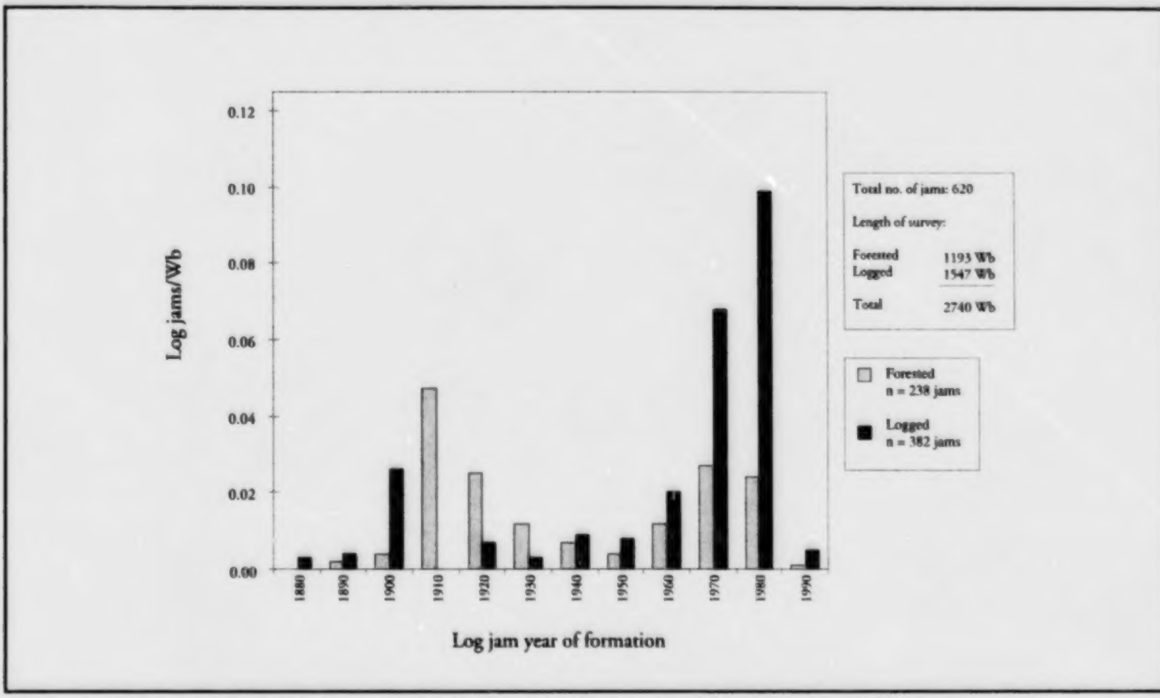

Figure 4. Large woody debris jam age distributions for forested and logged watershed streams.

Approximately equal amounts of debris in the jam are derived from upstream and the proximal riparian area. The riparian vegetation is introduced into the channel as a result of bank erosion initiated by the formation of the jam. Once the jam forms, it has a tendency to grow by the addition of wood from overbank areas.

Longitudinal channel surveys in the QCI study streams identified 620 LWD jams, including 238 and 382 LWD jams in forested and logged watersheds, respectively. The frequency of LWD jams through time (Fig. 4) indicates that the creation of LWD jams has been episodic over the past century (Hogan et al. 1995). The distribution is bi-modal, with the first peak centered near the turn of the 20th century, and the second in the 1970s. In comparison with landslide histories provided by Schwab (1998) and the corresponding meteorological histories provided by Septer and Schwab (1995), episodes of jam initiation correspond to landslide events trigged by severe rainstorms. The first peak on the histogram identifies an episode of LWD jam formation corresponding to storms that occurred in 1891 and 1917. The second peak identifies another episode of LWD jam formation, corresponding to storms that occurred in 1964, 1974, and 1978.

Both episodes of LWD jam formation were evident in all watersheds, regardless of land use. However, the frequency of LWD jam formation through time was mutually dependent on both watershed type (see Hogan et al. 1995, for details) and land use. Generally, the relative frequency of young jams increased as the watershed became smaller and steeper. In relatively large watersheds with predominately decoupled stream channels, mass wasting events rarely impacted the channel. Consequently, relatively few new LWD jams were created during episodes of watershed disturbance. As the connection between the hillslope and the stream channel becomes stronger, mass wasting events create LWD jams at an increasing rate, often destroying old LWD jams in the process. As the channel gradient becomes increasingly steep, the rate of jam production, with an increasing connection to the hillslope, may reach a critical point. For example, during a debris flow in a steep, coupled stream, the entire channel is scoured by debris passing completely through to the stream mouth from upslope source areas. Large woody debris jams that exist before such an event can be completely destroyed.

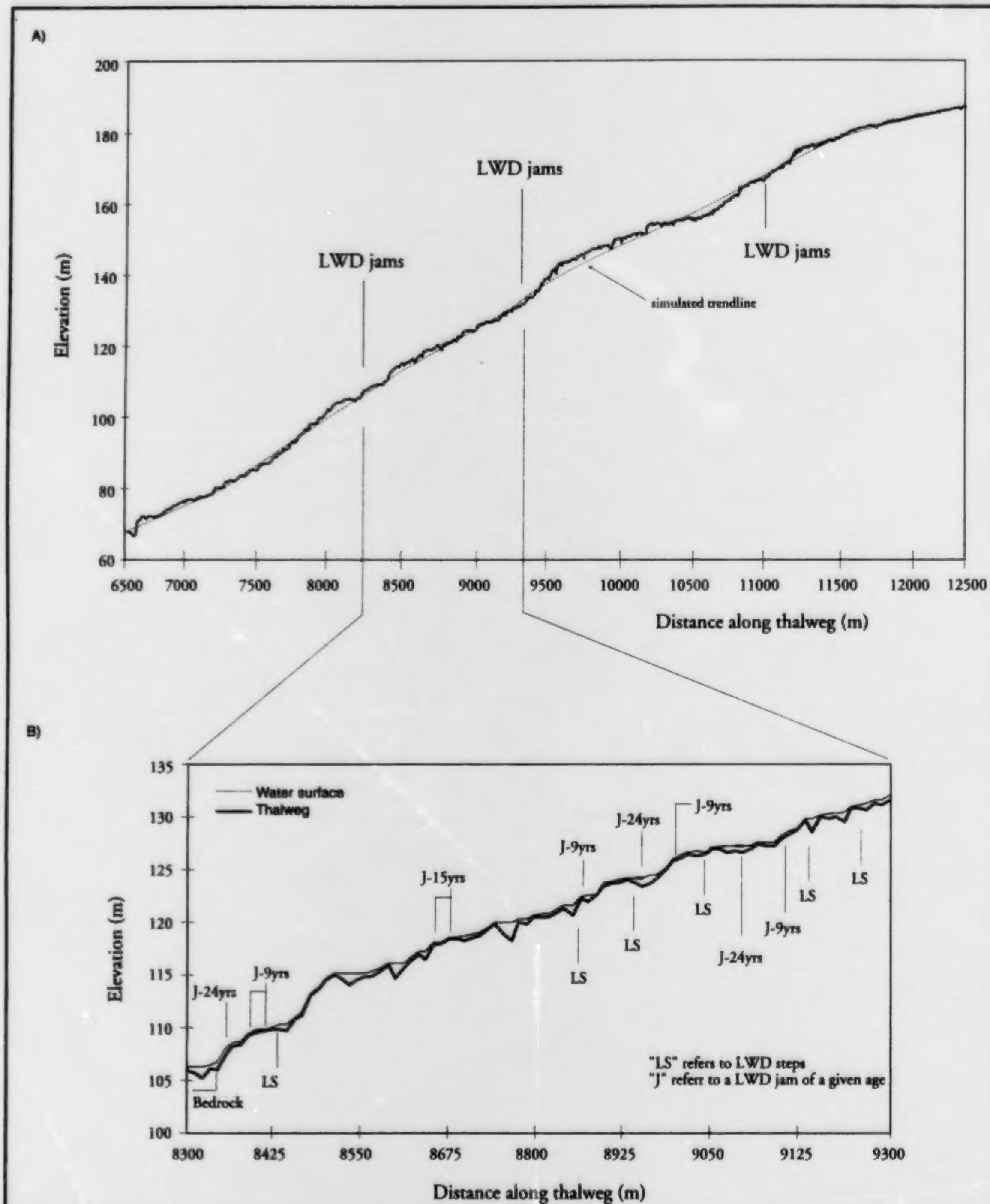

Figure 5. Longitudinal profile of a logged watershed stream (Riley Creek). A) Entire longitudinal survey (zones lying above the smooth line drawn through the data points—the smooth line is a second-order polynomial placed simply to make visual inspection easier). B) Detailed profile of selected section from A.

The influence of landuse on LWD jam formation is apparent by considering the impact of the 1978 storm. The large number of jams initiated in 1978 was due to the accelerated rate of landslide occurrence in logged watersheds. The lack of old jams (initiated in 1917) in logged watersheds is likely a result of their replacement by young jams initiated by the 1978 episode.

The frequency distribution of LWD jams through space increases slightly with logging. Longitudinal channel surveys identified 238 jams (0.22 jams/Wb) and 382 jams (0.26 jams/Wb) in forested and logged watersheds, respectively (Hogan et al. 1995). Generally, in forested watershed streams the main anchors of LWD jams are large root wads, previously existing jams, mid-channel islands, and bedrock knobs that constrict flow. However, in logged watershed streams, most jams develop on top or immediately upstream of older jams and do not, therefore, significantly alter jam frequency.

The Evolution of Channel Morphology

Research on the influence of LWD on channel morphology has concentrated on changes in the size, amount, and function of LWD pieces (e.g., Thomson 1991; Bisson et al. 1987; Hartman and Scrivener 1991; Hogan 1987). The results of such studies have been applied to forest management issues, and have proven beneficial in solving specific problems, such as the need for streamside buffers to provide a continued LWD source. Although the role of individual LWD pieces is important and influences morphology, LWD jams play a larger role in controlling the evolution of channel morphology.

Large woody debris jams have an important influence on the longitudinal profile of small coastal streams (Hogan 1989). This is seen by the extensive sediment accumulations upstream of the debris jams (Fig. 5). The spatial extent of debris jams and their influence on the longitudinal profile is evident in Figure 5B. Major sediment wedges are formed upstream of large jams or multiple jams that are closely spaced. Between the jams, the channel has a typical pool-riffle sequence (mean spacing distance of 4.3 Wb), and cobble-gravel textured bars. Individual LWD pieces, particularly debris steps, are important. Because jams control the largest features (particularly the extensive sedimentation upstream and degradation downstream of jams) they have the greatest influence on the overall channel. The individual debris pieces are important to channel morphology and fish habitat features at a smaller scale, primarily between the jams.

A Model of LWD Jam Evolution

Streams influenced by LWD are characterized by complex and diverse channel features. Hogan et al. (1998) show examples of stable CWH stream channels located in decoupled watersheds. Overall, the channel is diverse, with complex longitudinal and planemetric forms, and distinct, well defined pools and riffles (Hogan 1986). Pools, primarily lateral scour pools, account for almost 65% of the overall channel area. Considerable variation is also evident in channel width, as the channel alternates between narrow and wide reaches. Banks are commonly undercut and channel bars consist of cobble, gravel, and sand size materials. The LWD is prevalent, and frequently spans the channel from bank to bank. The predominant orientation of the debris pieces is either perpendicular or diagonal to the general alignment of the banks. Most of the debris have root wads attached to the log trunk.

Debris flows that enter the stream channel deposit large volumes of sediment and LWD into the channel, and reorganize existing pieces of LWD into jams. The development of a LWD jam following a debris flow in Carnation Creek (CC) is shown in Figure 6 (annual surveys have documented the short term channel adjustments to jam formation in Carnation Creek). The jam (its photograph is shown in Figure 7A) began to form in 1976, but was greatly enlarged in 1979 as a result of a large storm. The channel width upstream of the fully developed jam increased 6-fold and there was over 2 m net bed aggradation. All previously existing pools, riffles, and bars were completely destroyed.

Large woody debris jams are different than other in-channel blockages such as those caused by rockslides that create essentially permanent dams. The LWD jams begin to break down over time. The debris pieces rot, are broken into smaller sizes, and are moved by floods. The longevity of each jam influences its temporal role in controlling channel morphology, as the interruption of sediment transport decreases through time (Hogan 1989).

Figure 8 shows a cobble-gravel bed channel with two distinctly different sections upstream and downstream of a LWD jam. The debris located in the middle of the channel is of two ages and origins. The large debris on top of the debris cluster is a result of recent blow-down (logs mapped as above the channel bed, and with attached root wads located in the overbank zone). Because this material is above the bed, it has had very little influence on bed and bank sediment scour or deposition. Beneath the

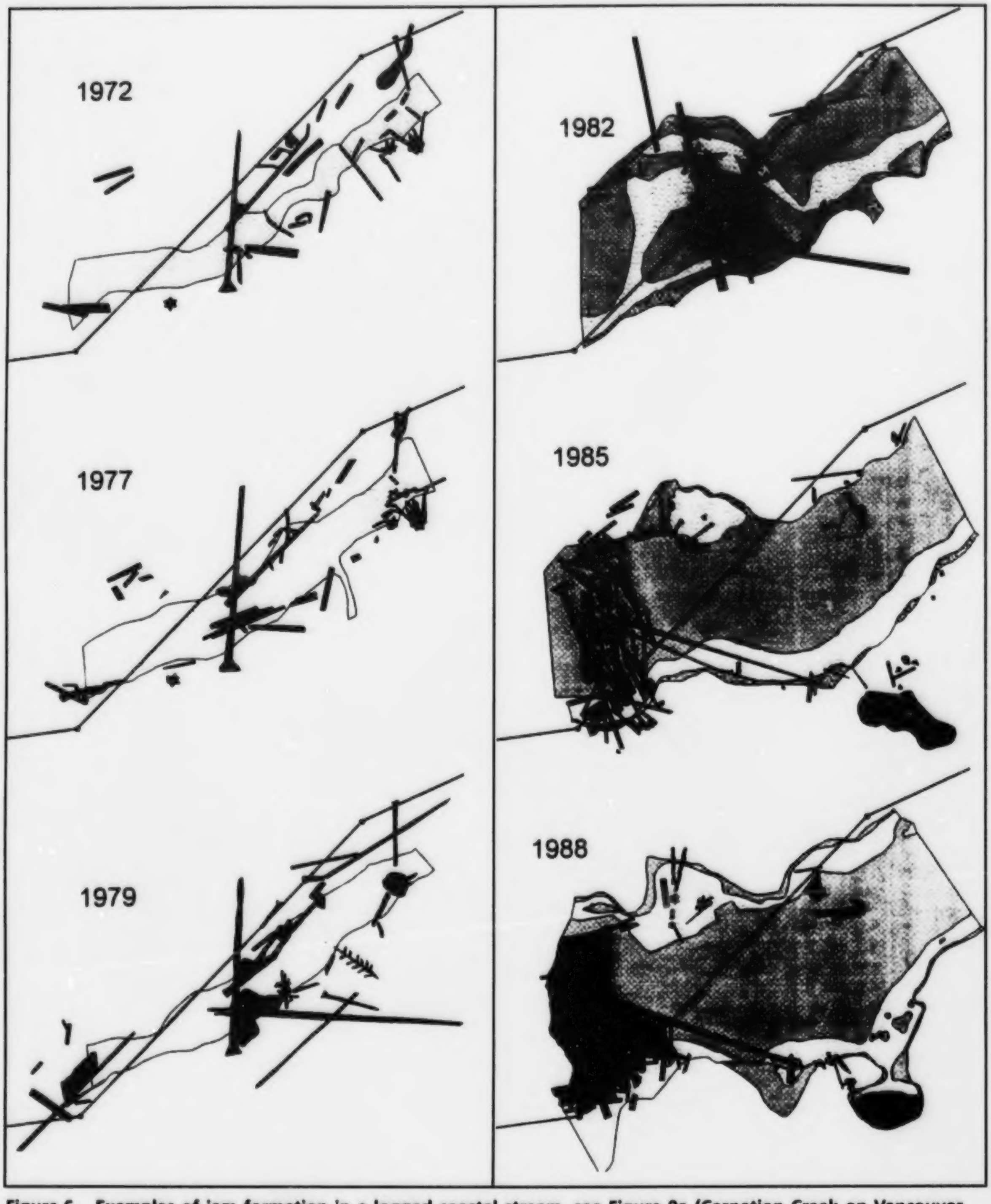

Figure 6. Examples of jam formation in a logged coastal stream, see Figure 9a (Carnation Creek on Vancouver Island, B.C. used because of annual surveys available). Note that survey hubs and lines are in the same location on each map.

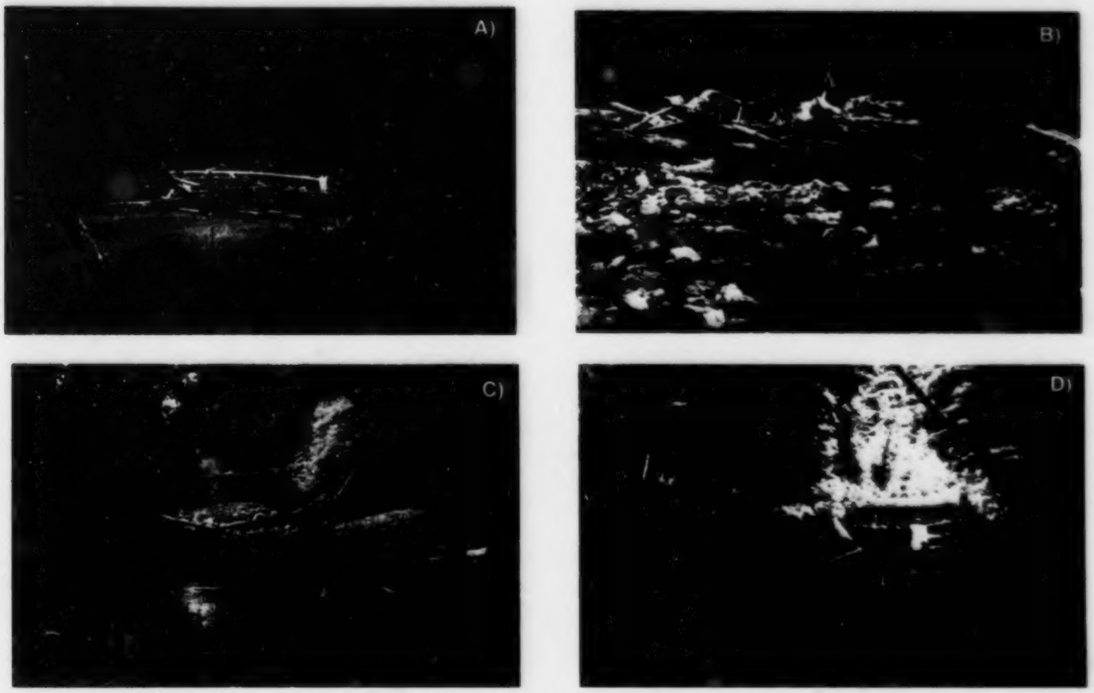

Figure 7. Large woody debris jams of different ages. A) Recently formed jam with extensive sediment wedge upstream of intact debris that effectively stops sediment transfer downstream. B) Moderately old jam (25 years since formation) showing evidence of downcutting of the sediment wedge as the jam deteriorates (no longer spans the channel). C) Old jam (50 years since formation) that has minimal contemporary influence on sediment trapping and scouring. D) Very old jam (300+ years since formation) with complex channel conditions (deep pools, log steps, under-cut banks, stable bars and back channels).

blown-down wood is an old debris jam. Nurse trees growing on in-stream debris had ages ranging from 32 to 45 years, indicating that the jams had been in place for several decades. The vegetated, undercut, and stable banks and islands attest to these ages. The remains of logs are also incorporated into the fluvial sediments making up the channel banks.

The downstream zone in Figure 8 extends from 0 to 180 m. It has a single channel with long, well defined pools and riffles, gravel textured side channel bars, and diagonally oriented LWD pieces that cause lateral and under-scour pools. Upstream of 180 m, the channel expands laterally, doubling in width, and has multiple channels with mid-channel bars and vegetated islands. Pool and riffle shapes are very different in the upstream section compared to the downstream, with increased lateral and vertical variability. Examples of old and very old log jams (300+ years since jam formation) are given in Figures 7C and D.

Large woody debris age and channel morphology are intricately linked. Therefore, the shift in LWD jam age distribution documented in Figure 4 will lead to a corresponding shift in expected channel morphology.

Hogan (1989) proposed a model of temporal and spatial adjustments of channel morphology in response to the development of LWD jams (Fig. 9). Initially, prior to the formation of a LWD jam, the channels are very morphologically complex with many of the features shown in Figure 8 (Fig. 9a). After the jam has been established, the channel undergoes fundamental changes. The most severe changes occur during the first decade (Fig. 9b). Recently formed jams act as effective sediment traps that cause bank erosion and increased widths, reduced gradients, and finer sediment textures upstream of the jam due to bed aggradation. During the second and third decades the jam begins to deteriorate, becoming a less effective sediment trap, and

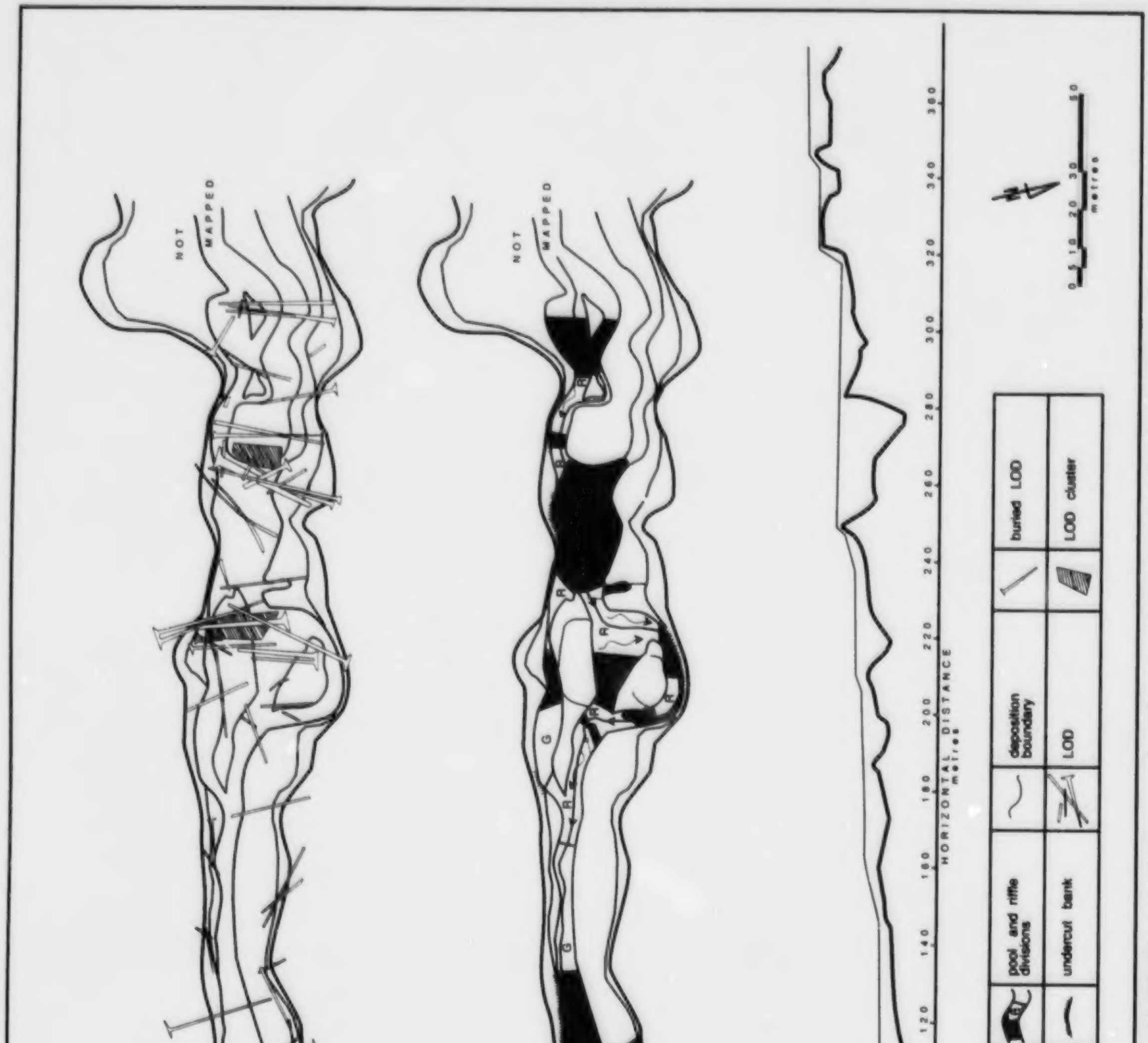

Figure 8. Channel morphology map showing an old (45-50 years) large woody debris jam (map legend included in Figure 6).

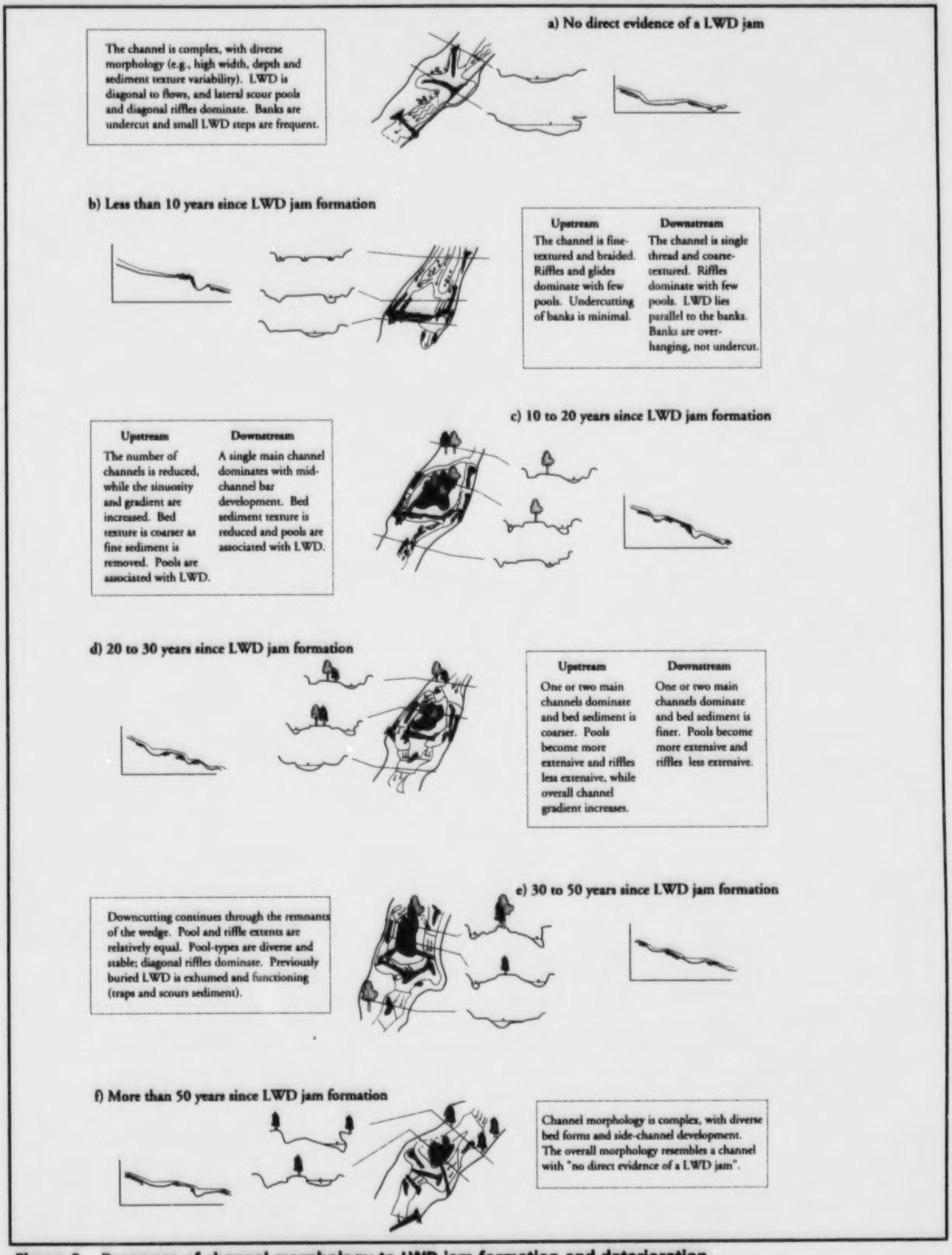

Figure 9. Response of channel morphology to LWD jam formation and deterioration.

the sediment supply to downstream zones increase. As a result, the upstream wedge is downcut, preferred channels are established, and riparian vegetation begins to colonize the bar and bank surfaces (Fig. 9c and d). Typically, after about 30 years the channel begins to resemble pre-jam formation conditions (Fig. 9e and f). Although remnants remain along the channel margin, and individual LWD pieces remain along the bed, after 50 years there is very little evidence of the original jam (e.g., Fig. 8). Many of the debris steps shown in Figure 5 are actually the final remains of ancient jams.

The Temporal Influence of LWD Jams on Channel Morphology

Field examples of channel changes associated with debris longevity are shown in Figure 10 (Hogan 1989). Debris jam locations and ages are shown in the longitudinal profiles to enable comparison of specific features associated with the various ages. In all cases, there is substantial channel aggradation upstream of the recently formed LWD jams. For example, at the 3400 m distance (Fig. 10a), the bank top and bar top surface graphs merge, indicating that the channel bed is at the same elevation as the bank top. The channel has completely filled with sediment, and the bar tops are elevated to the height of the bank tops. Degradation of the channel bed is also evident downstream of the young jams (3350–3400 m). As LWD jams age, more sediment is excavated. The bed upstream of the jam downcuts, and eroded sediment is transferred downstream. For example, there is relatively little aggradation between 2800 and 3000 m and from 3150 to 3350 m, although there are two large older jams present in these zones.

The volume of LWD in a jam also decreases with time (Fig. 10b) as individual pieces rot, or are broken into smaller pieces and removed by higher flows. As jams are reduced in size, the orientation of the remaining LWD shifts from perpendicular to parallel relative to the stream bank (Fig. 10c). This shift in orientation influences the sediment trapping ability of the debris, increasing the effectiveness of bed and bank scour through time (Hogan 1989).

Changes in channel width and sediment texture are also related to LWD jam age (Fig. 10d and e). In general, where the channel walls are not confined by bedrock there is an increase in channel width associated with young jams. There is a reduction in sediment size coinciding with the wider, aggraded, channel upstream of the young jam. In most instances the sediment texture is finer immediately upstream of a jam, and considerably coarser downstream. However, as LWD jams break apart, the width increases, and sediment sorting characteristics of young jams become progressively inconspicuous.

Large woody debris jams are spatially prevalent (Hogan et al. 1995). Considering all surveyed channels on the QCI, the median spacing is 2.85 and 2.30 Wb in forested and logged streams, respectively (Fig. 11a). The spacing is slightly longer in CC, averaging 3.7 Wb in the logged sections. This means that, on average, there is one debris jam every 2–4 channel widths (if Wb = 15 m, then 1 jam is found along every 30–60 m of channel). However, in the field, only the recently formed jams are obvious, so the spacing appears much longer than reported here. The spacing distance of young jams is much greater than for the total of all ages. For instance, in one QCI stream, approximately 50% of the young jams are spaced further than 14 Wb apart (Fig. 11b).

The Spatial Influence of Forest Management on Channel Evolution

Large woody debris jams are fundamental structural elements in the small coastal streams investigated in our studies. The recently formed jams alter channel morphology to the point that in-stream fish habitats are essentially destroyed. Over the course of 50 years the same LWD jam creates complex, diverse morphologies that are highly productive fish habitats. Therefore, the shift from an even distribution of young, moderate and old LWD jams, to predominately young LWD jams, constitutes a critical impact.

Hogan et al. (1995) identified two episodes of LWD jam formation that have occurred in the Queen Charlotte Islands in the last century (Fig. 4). The jam-forming magnitude of Episode I (storms in 1891 and 1917) is fairly similar in both forested and logged watersheds. (The peak rate of LWD jam formation is 0.0047 and 0.0026 jams/Wb/yr in forested and logged watersheds, respectively.) This is because the episode pre-dates logging i.e., the "logged" watersheds were unlogged at the time. However, the magnitude of Episode II (1964, 1974, and 1978 storms) is significantly greater in the logged watersheds. The peak rate of LWD jam formation is 0.0027 and 0.0099 jams/Wb/yr in forested and logged watersheds, respectively. Generally, both forested and logged watersheds have similar distributions of old jams, but logged watersheds have more new jams. Therefore, given the distribution of LWD jam frequency through time (Fig. 4), during a contemporary inspection of a stream in the Queen Charlotte

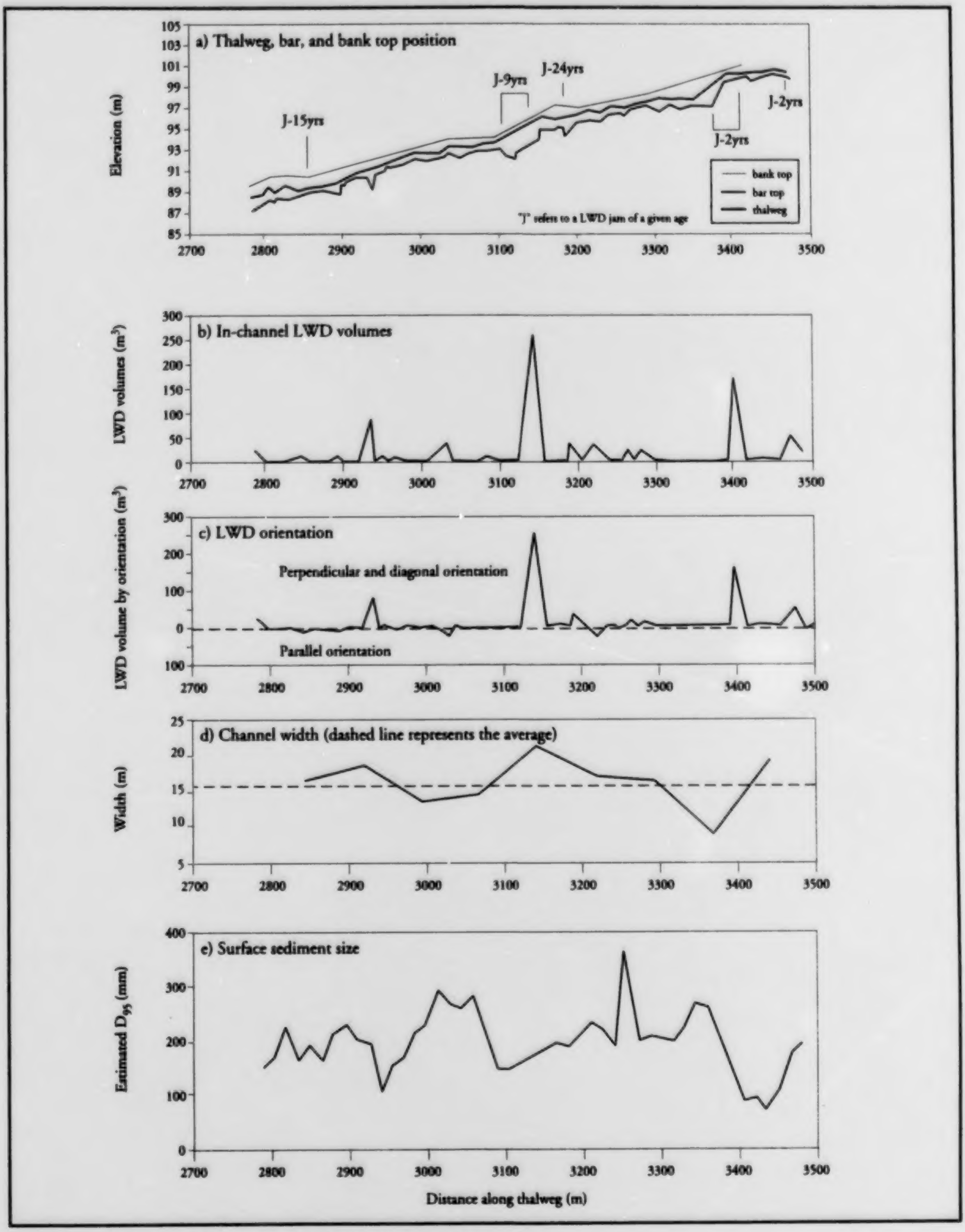

Figure 10. Channel changes associated with LWD jam longevity.

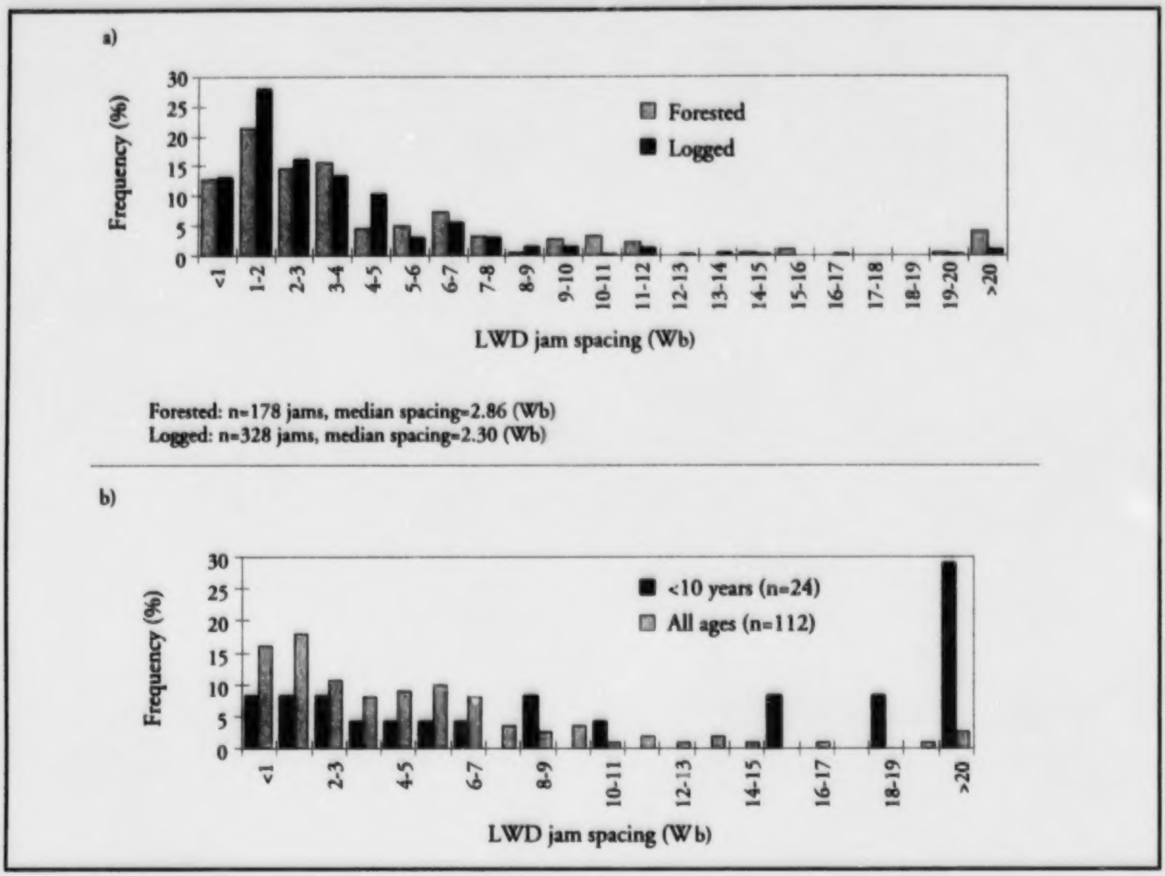

Figure 11. Spacing of LWD Jams. A) LWD jams in forested and logged streams, and B) recently formed LWD jams.

Islands, we would expect to find old and young channel characteristics (Figures 9a and b) equally distributed through space in forested watersheds. In logged watersheds, we would expect to find young channel characteristics nearly four times as often as the old morphologies.

Management Implications

Much of the channel diversity that characterizes coastal streams is a result of LWD jam formation and longevity. Over the long term, on the order of half a century, the complex channels and riparian zones that develop as a LWD jam deteriorates are highly productive fish habitats. However, the habitat conditions are very inhospitable for fish during the early phases of channel adjustment to jam formation. Spawning areas (riffles) are buried upstream of the jam or eroded downstream of the jam. Rearing pools are infilled, and egg incubation environments are smothered with fine textured sediments. Therefore, any interference with the natural evolution of stream channels by shifting the relative frequency of recently formed jams, constitutes a fundamental environmental impact.

In an old growth forested watershed, the natural rate of LWD jam formation is very low. Consequently, in a forested stream there will be a wide range of jam ages, and no one age will dominate the morphology. Although some age classes will be more prevalent because of the episodic nature of landslides, the range of ages produces a diverse mosaic of channel patterns that have rich habitat attributes.

In a logged watershed, the rate of LWD jam formation is accelerated. The nature of entire channel systems can be altered because the steep headwater streams receive proportionally more jam forming events, but the influence of these are transferred downstream into larger, lower gradient streams.

A troublesome legacy of past forest management practices in steep terrain is the severity of the environmental damage produced by relatively low magnitude-high frequency storm events. The 1978 storm on the Queen Charlotte Islands was not as intense as events occurring earlier in the century. Far more landslides occurred during 1978 than in earlier storms of the same or greater magnitude. Previous studies have confirmed that logging of unstable slopes accelerates the already high rate of landslide activity along much of coastal British Columbia. This leads to a corresponding increase in recently formed LWD jams, with all of the associated channel morphology and fish habitat changes. New management initiatives, particularly the British Columbia Forest Practices Code, will strive to minimize future environmental impacts in streams. However, the current recovery of stream channels to their pre-logging conditions is dependent on the time required (about 50 years) for a diverse array of LWD jam-ages to establish.

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Description of Selected Hydrology and Geomorphology Studies in the Stuart–Takla Experimental Watersheds

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Abstract

Four hydrology and geomorphology studies have been initiated in support of the biological research that drives the Stuart–Takla Fisheries/Forestry Interaction Project. Snow accumulation and melt differences between clear-cut areas and undisturbed forest are being investigated. Sub-basin monitoring may isolate the impact clear-cut logging has on flows. Sediment delivery from road-related surfaces is being measured on a synoptic basis in the Takla Lake area, and the efficacy of revegetation techniques in reducing erosion relative to untreated areas is being evaluated using runoff plots and erosion measurements. Groundwater upwelling, a possible mechanism accounting for high sockeye salmon egg to fry survival rates, is being evaluated using nests of piezometers.

Introduction

The Stuart–Takla Forestry Interaction Project was initiated in 1991 to study the impacts of logging on three highly productive sockeye salmon spawning streams about 100 km northwest of Ft. St. James, British Columbia. This paper describes four of the hydrology and geomorphology studies are being undertaken in support of the biological research that drives the project.

The three main Stuart–Takla watersheds included in the hydrology and geomorphology studies (Gluskie, Forfar, and O'Ne-ell Creeks) range in elevation from slightly below 700 m at their outlets to about 1980 m at the peaks of the Hogem Range, which forms the main drainage divide. Creek drainage area ranges from 37.4 km^2 to 73.3 km^2 (Cheong et al. 1995). Lower elevations, where main forestry haul roads have been built, are dominated by highly erodible glaciolacustrine sediments. It is expected that these sediments will have a significant impact on the study streams. The terrain and sediment sources of the Stuart–Takla watersheds have been described in detail by Ryder (1995). The mean annual precipitation is 50 cm.

Snow Hydrology

The study watersheds are hydrologically dominated by snow. The movement of sockeye fry out of their natal streams and into Takla Lake and the Middle River in the spring has been observed to be strongly influenced by the timing and magnitude of spring snowmelt events. The impact of logging on this process is of direct interest, since patterns of fry predation by resident species may be affected by

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hydrologic changes generated by forest removal (Macdonald et al. 1992). The hypothesis of interest is that clear-cut logging will result in a snowmelt regime that begins earlier, which will result in fry moving out of their streams earlier and over a longer period, generally at lower daily rates (Macdonald et al. 1998). This runs counter to the preferred survival strategy of overwhelming predators with large numbers over a short period, and matching fry migration with food availability in rearing areas (Burger 1991).

Research in southeastern British Columbia and elsewhere has shown that clearcuts tend to have higher depths and water equivalents of accumulated snow compared to undisturbed forest (Toews and Gluns 1986; Harr and Berris 1983). The timing of snowmelt in a watershed is altered by clear-cut logging, and the total amount of water derived from the snowpack is increased (Troendle and Leaf 1981). Trees planted in clear-cut areas must reach a critical size and density before they begin to behave like an old-growth forest in their interactions with snow, at which point the clearcut approaches the theoretical state of "hydrologic recovery." Hudson (1995) reported achievement of 90% hydrologic recovery with respect to snow under various coastal British Columbia forest regrowth conditions. Little information on hydrologic recovery is available for interior regions of British Columbia. The proportion of equivalent clear-cut area (ECA) in a watershed also influences the overall impact of snowmelt changes on streamflow. The extent of ECA required before snowmelt-induced changes in streamflow are detectable also varies with the topographic distribution of logging, the properties of the soils and surficial materials, and the accuracy of stream gauging.

One of the project's main goals is to extend findings to other areas in the interior region of British Columbia. Thus, it is important to document and understand how processes of snow accumulation and meltwater delivery to streams differ between clear-cut and the forested land, especially since this information is not generally available for north-central British Columbia.

Differences in snow accumulation–melt relationships between forested and clear-cut areas are being evaluated by:

1. recording snow depths with ultrasonic depth/height sensors,
2. measuring hourly precipitation using a standpipe precipitation gauge,
3. recording snow depth, water equivalent, and density via periodic manual snow surveys,
4. monitoring rates and timing of snowmelt using snow lysimeters,
5. monitoring wind speed with an anemometer to evaluate whether wind redistributes snow,
6. measuring soil temperatures with thermocouples, and
7. monitoring soil moisture content with reflectometers to assist in determining the proportion of snowmelt infiltrating the soil versus that which runs off the surface.

Standard snow surveys are being conducted monthly in the spring at high, medium, and low elevation sites in Forfar Creek watershed to establish relationships between elevation and snow depth, water equivalent, and density. These data will be combined with the clearcut snow hydrology results to provide some of the input necessary for snow modeling. It is expected that model parameters will also be derived using GIS analysis of digital terrain maps and spatially distributed land-use data. Snowmelt modeling will be an important tool in extending the project findings to the region as a whole.

Streams in the study watersheds are gauged by stage recorders supplemented by velocity profiles taken manually to obtain a stage–discharge relationship (rating curve). Discharge values are calculated using continuous stream-stage records. The merchantable timber resources in the study watersheds have resulted in logging plans that will produce an ECA of about 20%, mainly in lower elevation (i.e., lower snow accumulation) zones. The error associated with stage–discharge relationships and the amount and distribution of logging may, in combination, make detection of snow-related hydrologic impacts in the study streams difficult.

Paired Sub-Basin Studies

It will be valuable to isolate the streamflow and sediment impacts of logging by other means, especially if detection of hydrologic changes in the logged study watersheds is hampered by stream measurement uncertainty and relatively low clearcut proportions. The most effective method of focusing on hydrologic impacts from logging in the case of the Takla watersheds is through the use of paired sub-basins, where flows can be accurately measured. Two similar adjacent sub-basins, about 50 ha each, have been selected in the mid-elevation zone of the O'Ne-ell Creek watershed. Monitoring of hydrologic response and sediment delivery has begun and will continue for a pre-logging period of at least 5 years to characterize the behavior of each sub-basin under a range of climatic conditions. One of the sub-basins will then be entirely clear-cut logged while the other is retained as a forested control.

Ninety degree v-notch weirs have been installed on two adjacent small sub-basin stream channels just above their confluence. Discharge and temperature are measured every 30 minutes. Snow will be measured by both snow courses and automated ultrasonic snow depths. Snow lysimeters are to be installed in both watersheds for snowmelt measurement, and soil moisture will be measured to enable snowmelt routing investigations. Tipping bucket and manually monitored rain gauges have been installed to measure liquid phase precipitation during snow-free periods of the summer months. Reliance on datalogger-based data measurement and recording systems is dictated by the difficult logistics of the sub-basin locations, which are more than 5 km by trail from the nearest vehicle access. Manual measurements and on-the-ground equipment servicing are done periodically throughout the year. The snow hydrology work at low elevations, described above, will augment the mid-elevation sub-basin studies. Both data sets will be used to produce watershed process models for the snow-dominated systems of British Columbia's central interior region.

Bedload capture by the installed weirs is used to measure coarse-grained sediment transport from the sub-basin creeks. Automated methods of monitoring suspended sediment concentrations are under investigation, but not yet implemented.

Groundwater Upwelling

The presence or absence of warmer groundwater upwelling during winter into the egg and alevin bearing gravels of the study streams is key to understanding the biology and incubation environment of the early Stuart sockeye salmon. Hydrologic impacts from logging also have the potential to alter existing groundwater processes in and around fish bearing streams.

Nests of two or more piezometers are being used to measure vertical groundwater pressure gradients in order to establish whether upwelling groundwater is present in the Takla streams. The piezometer nests are installed immediately adjacent to the normally wetted portion of the channel. Some piezometers are automatically metered using pressure transducers connected to dataloggers; these are supplemented by a network of piezometer nests measured manually with a circuit closure device. Groundwater and stream temperatures are also being measured using thermocouples, connected to dataloggers.

Road-Related Sediment Sources

Forestry roads are a major source of fine sediment due to soil erosion from road surfaces, cut and fill slopes, and ditches (Anderson and Potts 1987). Stream crossings are a major point of delivery of these sediments to streams. A portion of these sediments settle into the gravel pore spaces once they have been transported into stream channels. Fish eggs can suffocate, and alevins may be entombed as a result of fine sediment additions to stream gravel (Moring 1982). Depending on their effectiveness, controls on road sediment production may significantly reduce logging impacts on fisheries streams. However, in order for erosion control measures to be optimized, the relative contributions of road-related erosion surfaces under differing environmental conditions need to be better understood.

Soil erosion from road-related surfaces is being calculated by carefully measuring the height of the soil surface after precipitation events and inferring soil erosion or deposition from surface height changes at each location. The device used for these height measurements is called a "rill meter." The rill meter consists of a portable reference crossbar, with measuring rod and ruler, that mounts on two permanently installed metal datum posts. Rill meter sites are distributed on a representative selection of roadcuts, for example, those with different slopes, aspects, and soil material types. Erosion and deposition in ditches is being measured in the same way using the rill meter.

Erosion plots on cutbank slopes are a second approach to studying sediment generation from roads. These involve collecting runoff samples to determine how effective two revegetation techniques (broadcasting vs. hydroseeding) are in controlling erosion, relative to an unseeded control. Soil losses from traveled road surfaces may also be measured by collection methods during runoff events. Coshocton wheels are being employed in erosion plot studies. The use of Coshocton wheels is a well-established method for sampling runoff and sediment from agricultural plots (Brankensiek et al. 1979); they have also been used in forestry-related research (Oyarzun and Pena 1995) . These devices consist of a metal disc with spiral flutings that is spun by the force of runoff water spilling from an attached flume; a slot elevated above the disc surface samples 1% of the total runoff on each rotation. Use of the rill meter on the erosion plots will enable cross calibration of the two measurement methods. Sampling and maintenance will be carried out between storms so sediment generation data from a

range of storm intensities will be obtained. These data will also provide required input into basin sediment budgets and be used for sediment modeling.

Summary

The four hydrology and geomorphology studies described here are in their early stages and have been presented to round out the picture of research activities currently underway in the Stuart–Takla Fisheries/Forestry Interaction Project.

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Suspended Sediment Regimes of Three Experimental Watersheds in the Takla Area of North-Central British Columbia

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Abstract

This study characterizes the baseline annual suspended sediment regimes and identifies sediment sources of three experimental watersheds with high fisheries values, with the objective of identifying any future forestry-related changes to the natural sediment budget. Pre-logging suspended sediment regimes were monitored annually near the mouth of three of the experimental watersheds since 1992 using both manual and continuous automated techniques. In each stream about 90–95% of the annual suspended sediment yield was transported in a 2-week period during spring peak flow. Very short pulses of sediment occurred during summer and fall storms. The annual sediment yield for each of the watersheds in 1995 was similar: 1.74, 1.60, and 2.35 $t{\cdot}km^{-2}{\cdot}yr^{-1}$ for Gluskie, Forfar, and Kynoch creeks, respectively. The greater yield from Kynoch Creek could be explained by the large lacustrine deposits in the lower reaches, which have a very high erosion potential. The sensitive soils located in all three watersheds, and the steep terrain, should provide a good test of the efficacy of the new British Columbia Forest Practices Code.

Introduction

With the exception of the Slim-Tumuch study east of Prince George (Slaney et al. 1977), the formal study of fish–forestry interactions has largely been ignored in north-central British Columbia. However, several large projects on the coast of British Columbia (B.C.) have studied such interactions; these include Carnation Creek (Chamberlin 1987) and the Queen Charlotte Islands (Poulin 1984). Although these projects were very successful, it is doubtful whether the knowledge gained from these projects is applicable to the northern interior of B.C., where the climate, topography, and geology are vastly different.

To gain a better understanding of fish–forestry interactions in the northern interior of B.C., the Stuart–Takla Fisheries/Forestry Interaction Program was initiated in 1990 (Macdonald et al. 1992). The project was initially instigated to provide input into the development of fish–forestry guidelines for the interior of B.C. However, the creation of the B.C. Forest Practices Code (FPC) (Legislative Assembly of B.C. 1994) superseded the development of the fish–forestry guidelines and changed the focus of the project. The focus is now directed toward testing the efficacy of the FPC rules and regulations, and providing information for the refinement of the regulations and standards. The history, detailed objectives,

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and experimental design of this program are described in Macdonald and Herunter (1998).

This paper describes the fine sediment budget component of the Stuart–Takla project and the initial results obtained studying the undisturbed watersheds (i.e., prior to forest harvesting). The effects of fine sediment on salmonids and salmonid habitats have been well documented in numerous publications (e.g., Hall 1984; Chevalier et al. 1984; Sorensen et al. 1977). The effects include: suffocation of eggs when sediment is deposited (Philips et al. 1975; Hartman and Scrivner 1990); filling of pools and interstitial spaces in the bed, thereby reducing available habitat (Saunders and Smith 1965; Bjornn et al. 1977); and direct mortalities when suspended sediment levels are extremely high (Noggle 1978).

Numerous studies have shown that forestry activities—such as yarding, road building, and road use—can significantly alter the natural sediment regime by increasing erosion and delivery of both fine and coarse sediments to the stream systems (Swanson et al. 1987; Johnson 1988; Cederholm et al. 1981). These increased rates of delivery of fine sediments have the potential to negatively impact salmonids and their habitats. The level of impacts depend on both the quantity of introduced sediment and the timing. Salmonids have different tolerance to suspended sediment and deposition (i.e., sedimentation) depending on the stage of their freshwater life cycles (Noggle 1978; Bisson and Bilby 1982). Consequently, it is important to describe both the timing of sediment yield and the quantity. Salmonids are often more affected by suspended sediment concentrations than total basin sediment yields. Consequently, it is necessary to investigate both total basin yields and peak sediment concentrations resulting from land-use activities.

The objectives of the study were to:

1) quantify impacts of forest harvesting on the fine sediment budgets of three experimental watersheds located in the northern interior of B.C., and
2) identify the sources of harvest-related changes to the fine sediment budget and provide operational recommendations to minimize future impacts.

Methods

The study uses a sediment budget approach as the primary method for assessing the impacts of forest harvesting on the basin sediment yield for three experimental basins. A drainage basin sediment budget identifies and quantifies the rates of production,

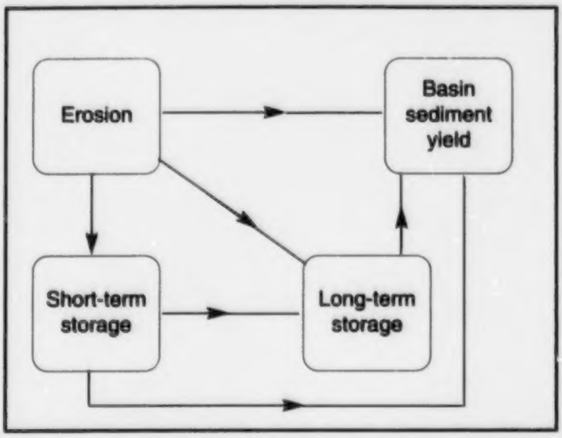

Figure 1. A flow chart of a conceptual sediment budget.

transport, and discharge of sediment from the watershed (Dietrich et al. 1982). Swanson and Fredrikson (1982) describe this methodology as: "providing a useful approach to holistic analysis of drainage basin functions." Such watershed scale information is necessary for evaluating the relative importance of harvesting-related sediment sources in the overall sediment budget, and how these sources can be minimized. The sediment budget can be expressed using the following equation:

$$I = O + S$$

where I represents the input of sediment, O represents the sediment yield, and S is the change in sediment storage over time. In this project the sediment budget was applied separately to the suspended sediment load of each of the three experimental watersheds. A conceptual sediment budget is presented in Figure 1.

Sutherland and Bryan (1991) provide an excellent synopsis of the results of sediment budget studies since the first field-based sediment budget study by Leopold et al. (1966). Most of these studies stress that there is no steady-state relationship between erosion and sediment yield (i.e., erosion does not equal sediment yield) (Megahan 1981). They clearly demonstrate that temporary storage of sediment is not negligible for small forested watersheds, and thus the assumption that S = 0 for short periods is inappropriate.

The following methodologies are being used to describe the three components of the sediment budget: 1) sediment source inventory and mapping (erosion sources); 2) stream channel inventory (storage

elements); and 3) suspended sediment and discharge measurements (basin sediment yield). A short description of each of the methodologies is provided below.

Sediment Source Inventory and Mapping

This work was completed and described by June Ryder and Associates under contract to the B.C. Forest Service (Ryder 1995). A brief summary of the methodology follows. Preliminary terrain mapping was carried out by air photo interpretation of 1:15 000-scale color air photos. Field checking was carried out with the objective of checking the air-photo mapping, particularly the surficial material described in each polygon, and to make observations of current sediment sources. Following field work, mapping from air photos was corrected and checked for consistency, and polygon boundaries were transferred to 1:20 000-scale TRIM (Terrain Resource Information Management) maps. The sediment source maps were prepared by interpretations of terrain map information supplemented by observations made during an overview flight of the three creeks.

Stream Channel Inventories

A reconnaissance stream channel survey was conducted in the lower 3 to 5 km of each of the three study streams (Gluskie, Forfar, and Kynoch creeks). The following data were collected every 50 m along the thalweg or at any location where there were major morphological changes: stream width; D90 of the fluvial sediments; stream gradient; observations and photographs of in-stream and streambank sediment sources; and description of morphological characteristics.

Suspended Sediment and Discharge Measurements

Monitoring of fine suspended sediment levels was conducted at one location on each of the three creeks. To avoid road effects, the sites were located upstream of the only road crossings in the watersheds. The Kynoch Creek site was located above the confluence of Tsitsutl Creek so as to eliminate its influence on discharge and suspended sediment delivery (Fig. 2).

The area of Kynoch Creek watershed above the sampling site is 38.5 km^2, excluding the Tsitsutl Creek watershed. The watershed areas above the sampling sites for Forfar and Gluskie creeks are 37.4 and 49.3 km^2, respectively. Concentrations of fine suspended sediment were measured using a combination of three techniques.

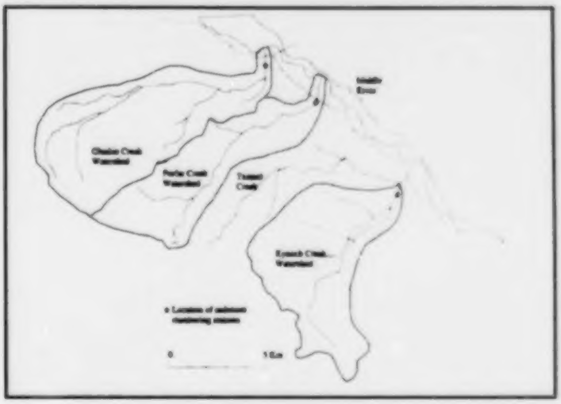

Figure 2. Location of the suspended sediment stations in the three experimental watersheds.

1) An optical back scatter meter (OBS) was used to collect continuous measurements of suspended sediments. This electronic device consists of a high-intensity infrared (IR) emitting diode, a detector (four photodiodes), and a linear, solid-state temperature transducer. The OBS measures turbidity and suspended solid concentrations by detecting IR radiation scattered from suspended matter. The OBS sensors can be calibrated to measure turbidity or suspended solids directly (D & A Instrument Company 1991). The OBS was connected to a Unidata data-logger to collect hourly average, maximum, and minimum readings.
2) An American Sigma 800 SL portable pump sampler collected discrete 1-L water samples at the same location as the OBS sensor. It was connected to the data logger and was activated by pre-set levels of suspended sediment measured by the sensor. Using this technique, the pump sampler only collected samples when the sediment concentrations in the stream reached a certain level (e.g., 10, 20, 30, 40, 50, 70, and >90 ppm).
3) During every site visit, three depth-integrated manual samples were taken across the stream at regular intervals using a DH48 sampler. These samples were used to assess whether the OBS readings represent the average for the stream cross section.

The pump samples were used to establish a relationship between the OBS readings and suspended sediment concentrations. The total basin sediment yields were calculated using the hourly

Table 1. Number of actual sediment sources in each of the experimental watersheds

Watershed	Small sediment source (2- to 10-m wide)	Moderate sediment source (10- to 40-m wide)	Large sediment source (>40 m wide)
Gluskie Creek	23	10	3
Forfar Creek	12	9	4
Kynoch Creek	7	1	2

Table 2. Percentage of each of the experimental watersheds in each of the potential sediment sources classes

Watershed	Very high sensitivity (%)	High sensitivity (%)	Moderate sensitivity (%)
Gluskie Creek	0	45	55
Forfar Creek	10	50	40
Kynoch Creek	35	20	45

OBS suspended sediment concentration data and the hourly stream discharge records. The product of these two measurements provided hourly sediment yields for the whole basin. Hourly results were summed to provide annual or seasonal yields. This technique provided a direct measure of suspended sediment yields without the need to develop sediment rating curves.

The methodology provides excellent results when all the instrumentation is working well, but numerous problems occur when using an optical device such as the OBS (Jordan 1996). Any obstruction of the infrared beam, aside from suspended sediment, renders the data invalid. Undesirable obstructions can include floating leaves, fish, and bio-fouling debris. When such obstructions occur, the pump samples become extremely valuable for identifying the erroneous OBS data and for cleaning up the data. Missing data can be filled in using a combination of the pump sample data and correlation to data from the two other streams.

Results and Discussions

Sediment Source Mapping

The sediment source maps (Ryder 1995) provided two kinds of information: actual sediment sources (e.g., recent landslides), and sensitive terrain or potential sediment sources. Actual sediment sources are point sources where sediment has been mobilized by landslides, chiefly debris slides and debris flows in till, glaciofluvial and glaciolacustrine sediments, and colluvium. All sediment sources that appeared to deliver fine sediments to streams were depicted on sediment source maps. These included streambank sites and scars on slopes that drained directly to the creeks or to narrow floodplains.

Relatively few actual sediment sources were identified and mapped in these watersheds. Table 1 provides the counts of these for each of the three size categories mapped.

Potential sediment sources are mapped polygons that have been designated to have significant erosion potential and/or are highly susceptible to slope failure. They have some or all of the following characteristics: fine-textured sediments; moderate to steep slopes; and wet soils. They are located adjacent to streams or tributaries, or in upslope positions from which sediment could easily reach the creeks. Three classes of sensitivity were assigned according to combinations of these characteristics (Ryder 1995). Very high sensitivity was assigned to steep slopes and to wet slopes underlain by glaciolacustrine sediments. High sensitivity was assigned to all other areas of glaciolacustrine sediments, to particularly vulnerable areas underlain by till, glaciofluvial sediments, and colluvium, and to slopes where sensitivity was demonstrated by recent occurrence of debris flows. Moderate sensitivity was assigned to other wet areas on till, to steep slopes on colluvium that drain directly to streams, and to stream-side scarps mapped as bedrock and colluvium.

Table 2 shows the relative importance of potential sediment sources in each of the watersheds (expressed as a percentage of all of the potential sediment sources mapped).

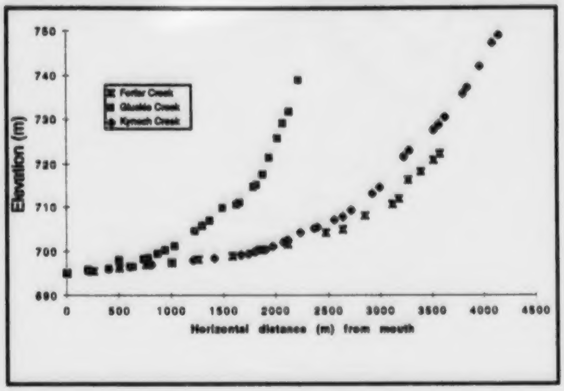

Figure 3. Longitudinal profile of the lower reaches of the three experimental creeks.

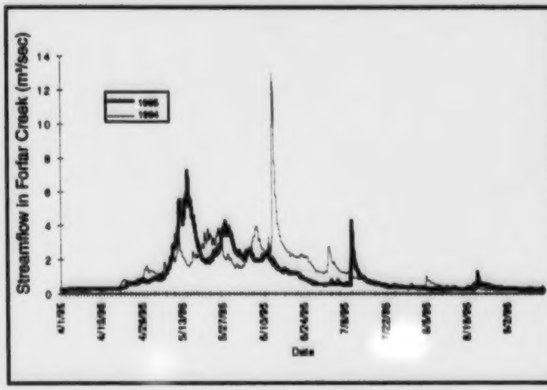

Figure 4. Stream discharge or Forfar Creek for 1994 and 1995.

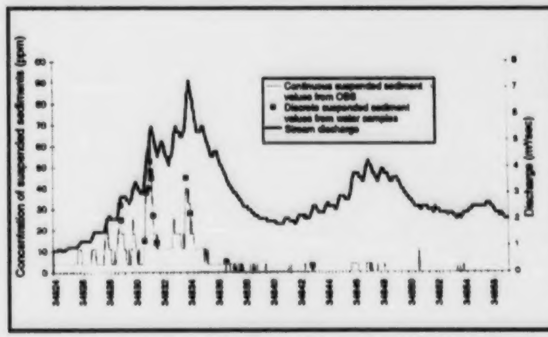

Figure 5. Spring runoff and suspended sediment concentrations in Forfar Creek during 1995.

Stream Surveys

Stream profiles of the lower reaches of the three watersheds were quite different, with Gluskie Creek being the steepest, and Kynoch and Forfar creeks having a much longer reach of low gradient (Fig. 3). However, the general structure of the creeks was similar: all creeks had a well-defined riffle-pool morphology at gradients below 3% with functioning large woody debris (LWD) characteristics. At gradients between 3 and 5% the streams tended more toward a cascade-pool morphology; at gradients greater than 5% the tendency was toward a step-pool morphology. In all creeks there were numerous large deposits of fine to coarse sands (0.05 to 2.0 mm) that served as an instream sediment source when streamflows were high. Streambanks themselves provided a large source of potential sediment for channels migrating across their floodplains. The numerous small sources that were evident during the channel surveys were recorded on field forms.

Fine Sediment Yield

The sediment yield results presented here are for 1995 only, as the analysis of the 1994 data is not complete. An attempt was made by Cheong et al. (1995) to develop sediment rating curves using the 1992 and 1993 manual data. However, insufficient data and the high variability of the data made the rating curves unreliable; consequently, sediment yields were not calculated for these 2 years.

Data obtained from Forfar Creek illustrate the typical characteristics of hydrograph and sediment yield curves for the three experimental watersheds. Figure 4 shows streamflows from 1 April to 15 September (1994 and 1995) in Forfar Creek. Flows before 1 April and after mid-September are generally less than 0.2 $m^3 \cdot sec^{-1}$, well below sediment transporting flows in these watersheds. This hydrograph is typical of northern interior watersheds where peak flows and the large majority of the annual water yields occur between early May and mid-June as a result of spring snowmelt and rain-on-snow events. Infrequent, large summer and fall rain events that occur about two to four times per year can cause high sediment transporting flows lasting from a few hours to a few days.

Figure 5 shows sediment concentrations and streamflow for the main sediment-producing event of 1995 in Forfar Creek. This type of response pattern is typical of all three experimental watersheds, where the rising limb of the first annual peak streamflow carries much more sediment than similar flows during the falling limb or rise of later peaks. After the initial large streamflow peak, sediment concentrations

Table 3. Suspended sediment yields for 1995 for each of the three experimental watersheds

Watershed	Annual suspended sediment yield ($t \cdot km^{-2}$)
Gluskie Creek	1.74
Forfar Creek	1.60
Kynoch Creek	2.35

are much lower for similar-sized events. This suggests that fine suspended sediments in the Takla Lake watersheds are supply limited, and that it may take several months for the supply to build up again before it is large enough to raise the suspended sediment levels above ten to twenty parts per million (ppm). Figure 5 also shows the very good relationship obtained between the pump sampler data and the electronic OBS data.

Total annual yields of fine sediment during 1995 were calculated for each of the three watersheds using suspended sediment concentrations provided by the OBS and the stream discharge data. Forfar Creek provided a nearly flawless set of OBS data in1995. Unfortunately, this was not the case for Gluskie and Kynoch creeks. Some OBS measurements had to be rejected because of random problems with obstruction of the infrared beam caused by floating litter and debris. The missing OBS data were reconstructed using the pump sampler data and regression equations based on the Forfar Creek results. Table 3 provides annual suspended sediment yields for each of the creeks in 1995.

Figure 6 shows detailed sediment yields from Forfar Creek during the 1995 spring freshet. This figure highlights two important characteristics of sediment yields in the experimental watersheds:

1) Most of the annual yield (i.e., 97%) was delivered in a relatively short 1-month period, with the majority of the yield being delivered in a very short period of just a few days (i.e., 67% in 4 days); and
2) During the first rise in the hydrograph, sediment yields were sensitive to even small diurnal fluctuations resulting from daily snowmelt. After the initial streamflow peak, sediment yields barely responded to a substantial rise in streamflows (e.g., 21 May to 6 of June), suggesting a limit in sediment availability.

During 1995 only 3% of the annual fine sediment yield for Forfar Creek was generated outside of the snowmelt period during short-duration, high-intensity rainfall events (Fig. 7). This characteristic was typical for all three watersheds (Fig. 8).

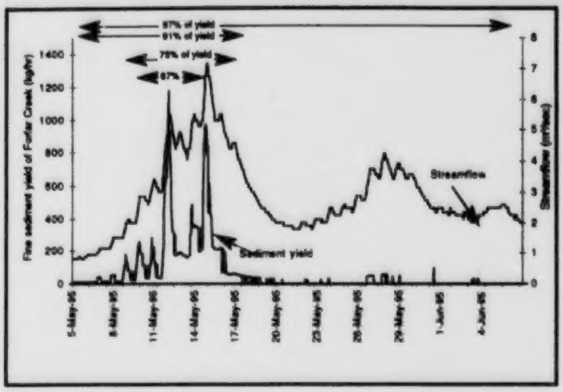

Figure 6. Suspended sediment yields and stream discharge for Forfar Creek during the spring of 1995.

It is interesting to note that the sediment source surveys presented in Tables 1 and 2 provide some conflicting information relative to the sediment yield data. Although Gluskie Creek had a much higher number of actual sediment sources than did the two other watersheds, these sediment sources were probably much less active than those along Kynoch Creek because the bedrock geology is less erosive (i.e., quartz diorite and granite vs. argillite and limestone). The small landslide scars along Gluskie Creek are generally on coarse materials or even down to shallow bedrock; this may explain the inconsistency. Another explanation could be derived from the potential sediment sources data (Table 2), which indicate that Kynoch Creek has a very large percentage of its streamside areas in the very high sensitivity class, and this may actually be a more important source of fine sediments than the old landslide scars. This makes sense considering that much of the sediment comes from direct streambank erosion and direct channel activity rather then directly from point sources. Another important consideration is that, with only 1 year of data, it may be premature to actually state that there is an actual difference in sediment yields between the three watersheds.

Conclusions/Implications

These preliminary results from the Takla experimental watersheds suggests that the watersheds have very low annual suspended sediment yields compared to other watersheds in B.C. and the Pacific Northwest. Church et al. (1989) report yields from 3 to 70 $t \cdot km^{-2}$ for small non-glacierized watersheds in B.C., which ranged in size from 10 to 200 km^2. Jordan

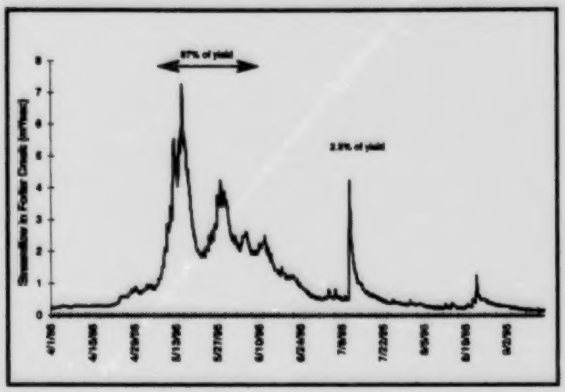

Figure 7. Annual distribution of suspended sediment yields for Forfar Creek in 1995.

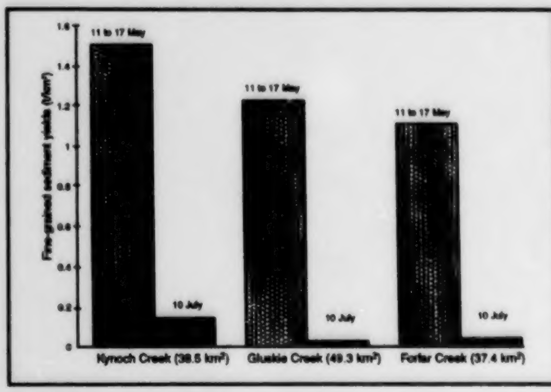

Figure 8. Specific fine-grained sediment yields for each of the three experimental watersheds in 1995.

(1996) reports yields of 1.3 to 4.1 t·km^{-2} for a small, unlogged watershed (15.0 km^2) and 2.2. to 8.4 t·km^{-2} for an adjacent similar-sized, roaded and logged watershed in southeast B.C. Suspended sediment yields at Carnation Creek (10 km^2), located on the west coast of Vancouver Island, B.C., ranged between 13 and 42 t·km^{-2} annually for the pre-logging period between 1973 and 1977 (Tassone 1987).

Patric et al. (1984) summarized sediment yield data from small, forested watersheds and larger watersheds of mixed land use in the United States. The average sediment yield for 26 small, forested watersheds on the Pacific Coast was 393 t·km^{-2} annually, with a range varying from 4 to 4356 t·km^{-2} annually. In the H.J Andrews Experimental Forest, located in the western Cascade Range, Oregon, pre-harvesting suspended sediment yields averaged 14 t·km^{-2} annually for watershed 1 (0.96 km^2) and 16 t·km^{-2} annually, for watershed 2 (0.60 km^2) (Grant and Wolff 1991).

These data from B.C. and the Pacific Northwest clearly show that specific fine sediment yields are relatively low for the Takla experimental watersheds. This may be attributed to several characteristics of these watersheds: 1) they are undisturbed by land-use activities—consequently, most of the sediment comes from channel erosion rather than slope erosion, which characteristically yields less sediment (Walling and Webb 1996); 2) there are only a few sediment-transporting flows per year, the biggest one being associated with spring snowmelt; 3) they have few active natural sediment sources such as landslides and gullies that deliver sediment directly to the main channel and its tributaries, and 4) the granitic bedrock geology has a relatively low erodibility.

About 95% of the annual sediment yields occurs in a 1-month period in the spring during snowmelt. The specific sediment yields—the quantity of sediment yielded per unit area of land surface—are similar for all three basins; however, Kynoch Creek is slightly higher, probably because of the greater abundance of lacustrine deposits (Table 3).

Although the present sediment yields are low, the potential for increased sediment delivery as a result of poor harvesting practices are high because: 1) there is an abundance of highly erodible materials in the lower reaches of the three watersheds, especially in Kynoch Creek; and 2) much of the merchantable timber is located on steep slopes with high drainage densities.

In conclusion, these watersheds should provide a good test of the efficacy of the new FPC of B.C. in protecting the fisheries resource from increased delivery of fine sediments to the stream channels.

Acknowledgments

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Influence of Aggregation on Storage of Fine-Grained Sediments in Salmon-Bearing Streams

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Abstract

An assessment of the role of aggregation of fine-grained sediment in salmon-bearing streams was undertaken as part of the larger in-stream processes component of the Stuart–Takla Fisheries/Forestry Interaction program. Fine-grained sediment (<63 μm) in suspension moves not only as single-grained particles but also as aggregates of fines that are held together by physical, chemical, and biological forces. The aggregates have different settling properties than do individual clay and silt particles, and they can potentially be stored in channels, on the bed surface, and within the gravel matrix. The hydrodynamic models that predict sediment transport in gravel bed streams do not account for these altered sizes. Consequently, the models mistakenly predict quicker flushing of fine sediments during extreme loading events (debris flows, bank slumping, and roadside erosion). This preliminary study of *in situ* particle size using underwater photography and image analysis techniques has indicated that aggregates in excess of 1200 μm diameter occur in these headwater salmon-bearing streams. The aggregates, comprising inorganic particles that do not exceed 130 μm are stored within the gravel matrix of the channel bed and within the sand matrix of the deltas formed at the mouths of the creeks. While the watersheds are currently limited by the sediment supply, future forest-harvesting activities could alter this balance, and much larger loads may be delivered and stored in their streams.

Petticrew, E.L. 1998. Influence of aggregation on storage of fine-grained sediments in salmon- bearing streams. Pages 241–247 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

The movement of sediment through a watershed depends on both hydrologic and geomorphic processes. Together, they regulate sediment erosion, transport, deposition, and resuspension, which, within the stream channel, has the potential to affect fish habitat (Petticrew 1996). Forest harvesting practices help determine the response of this coupling of hydrologic and geomorphic processes in headwater salmon-bearing streams.

An extremely important characteristic of the watershed that is modified by harvesting activities is the supply of sediments. Several studies within the larger, multi-disciplinary Stuart–Takla Fisheries/Forestry Interaction Program currently involve the collection of background data in order to differentiate the effects of forest-harvesting techniques on sediment supply and delivery to these highly productive salmon streams (Macdonald and Herunter 1998).

While surficial materials are generally abundant in glacially modified landscapes, the pre-harvest data indicate restricted availability of these sediments for erosion and transport in these systems (Beaudry 1998), which are supply limited. Future forest harvesting in these watersheds will modify variables such as vegetation cover and the timing and quantity of streamflow, which is expected to change the availability of sediment supply and drastically modify sediment routing in the system.

The forested watersheds of the Stuart–Takla area in the northern interior of British Columbia (B.C.) are underlain by extensive glaciolacustrine deposits (Ryder 1995) that are composed of high concentrations of silts and clays (<63 μm), hereafter referred to as fine-grained sediment. When roads are built and the forests are harvested from these watersheds, the probability of increased sediment loads being delivered to the streams is high (Bilby 1985; Bilby et al. 1989). As the streams in the Stuart–Takla study are high-quality salmon habitat (Macdonald and Herunter 1998), it is important to know how the processes of fine-grained sediment delivery and removal are regulated so that we can evaluate their storage time in the system.

Currently, hydrodynamic models indicate that fine-grained sediments should be carried quickly out of the system because they are moving as individual micron-sized particles that have very slow settling speeds. Evidence from many aquatic systems, both marine and freshwater, indicate that the particles aggregate, becoming larger and less dense, which then modifies their transport and erodibilty (Kranck et al. 1993; Droppo and Ongley 1994). If fine-grained particles are entering the systems as aggregates or flocculating in the streams as a function of the chemical or biological conditions (Muschenheim et al. 1989), the systems response to a large delivery of fines from slope failures, roadside erosion, or debris flows could be quite different from that predicted. The objectives of this work were to: 1) measure the size and concentration of suspended sediments to determine if the particles are transported as aggregates in salmon-bearing headwater streams; and 2) to determine if these aggregated fines are stored in river-bed gravels and river-mouth deltas.

Methods

Gluskie, Forfar, and O'Ne-ell creeks were sampled in August 1994 following the peak of sockeye salmon (*Oncorhynchus nerka*) spawning. Each of these creeks exhibit similar watershed size and channel slope structure. The last 2–3 km of the creeks cuts through a lowland area that is underlain by a belt of fine-grained glaciolacustrine sediment a few metres thick (Ryder 1995). The lower stream reaches exhibit low gradients (0.5–2.0%) and gravel sizes favorable for spawning. Each stream supports an extensive sockeye salmon spawning stock (Macdonald et al. 1992). Forfar and O'Ne-ell creeks have small, well-formed classic deltas where they empty into the Middle River, while Gluskie Creek has a smaller, less well developed sedimentary structure where the creek empties into Takla Lake.

A stream cross-section in the lower depositional zone of each watershed was selected. Sites were designated in regions where sockeye salmon redds were abundant, as it indicated quality spawning habitat. A stationary underwater silhouette camera was used to photograph the particles suspended in the water column. While it was securely positioned on the gravel bed, photographs of a water column 7.4 cm in diameter and 4.0 cm wide were taken at 2-s intervals. Timed water samples, collected for analysis of suspended sediments, were taken behind the aperture of the underwater camera. Current velocities were measured using a Marsh-McBirney current meter.

In each creek a series of photographs and water samples were collected during the undisturbed low flow conditions experienced in late August. A second set of samples was taken following a staged disturbance upstream. In an attempt to mimic the spawning effects of sockeye salmon, the gravels 4–5 m upstream of the camera site were artificially disturbed by a field assistant who reworked the bottom

gravels to a depth of several centimetres with his feet. The sediments resuspended by this disturbance were photographed and sampled; they referred to as post-disturbance samples.

Scuba-divers collected hand-held, plexiglass-encased sediment cores (approximately 30 cm in length, 6.0 cm in diameter) from the delta plains, slopes, and prodeltas at the mouths of each creek. Visual observation of the predominantly sandy cores allowed for the recognition of textural changes along their length. Regions of the core where strata breaks were exhibited (apparent fining of sediment) were extruded and subsampled. The core subsections were then transferred to the laboratory where they were processed for Coulter size fractionation (Milligan and Kranck 1991). This process includes the removal of organic matter using 35% hydrogen peroxide.

The water samples were filtered through triplicate, pre-weighed 8-μm SCWP Millipore cellulose-acetate filters. Suspended particulate matter (SPM) was determined gravimetrically on the dried filters and reported in $mg \cdot L^{-1}$. The weighed, dried filters were ashed in a low-temperature asher (<60°C) and wet digested with an excess of 35% H_2O_2 before analysis on a Coulter counter (Milligan and Kranck 1991).

A Coulter Multisizer IIE was used to determine the constituent or disaggregated inorganic grain size distribution of both the cored and filtered sediments. Results are expressed as a volume/volume concentration in ppm and are plotted as smoothed histograms of log concentration vs. log diameter (Milligan and Kranck 1991).

The *in situ* size distribution and concentration of aggregated particles were obtained by image analysis of the photo negatives obtained from the silhouette camera (Milligan 1995). The images were transferred to CD-ROM and imported into Jandel Scientific's Mocha image analysis program. The equivalent spherical diameter of the detected flocs was determined, and the particles were counted and grouped into size classes that correspond to the class intervals from the Coulter counter. They were plotted in the same fashion and on the same plots as the constituent particles. In the configuration used, the Multisizer has a lower detection limit of 0.63 μm and an upper detection limit of 1200 μm, while the floc camera has a lower detection limit of approximately 100 μm.

Results

The artificial disturbance of the salmon redd gravel beds resulted in an increase in both the concentration of suspended sediments and the size of particles transported. A staged disturbance of channel gravels would be expected to increase suspended sediment concentrations and maximum particle size in suspension, as the stored sands and fines would be resuspended. In each creek, post-disturbance suspended sediment concentrations increased by an order of magnitude (Fig. 1). The maximum grain size of the photographed suspended sediments also shows an order of magnitude increase following the disturbance of the gravel bed (Fig. 2). The constituent particles, which are the sediments sampled in the water column immediately after being photographed (but are disaggregated in the laboratory), did not exhibit a large change in maximum size following disturbance. The photographed particles that were released from channel storage from gravel bed disturbance are as large as 1290 μm. Coulter analysis indicates that these large aggregates are made up of the smaller constituent materials, which do not exceed 128 μm.

Fine-grained material is stored in channel gravels and is also observed in the delta sediments at the mouths of the creeks. Figure 3 portrays several samples taken from O'Ne-ell Creek watershed and includes a delta core sample that indicates the presence of fines in the predominantly sandy material. The sediment spectra of the roadside gully sample is rich in fines (mode about 8 μm). This extensive and deep glaciolacustrine deposit is a major sediment source in the lower portion of the watersheds; it delivers sediment to the creeks during runoff events such as spring melt and rainstorms. Grain size spectra of the delta cores from different water depths and different depths along the core (Fig. 4) also indicate that fines are being stored at positions along the delta slope (6- and 10-m samples) and in the prodelta (16-m sample). The evidence of similar concentrations and spectra slopes of fines in the deeper (20-cm) section of the cores indicates that the processes of aggregation and storage have been operating in the system in the past.

Discussion

Sizing of the suspended sediments by *in situ* photography combined with laboratory analysis of the inorganic grains, which comprise the suspended material, allows us to determine that aggregates of

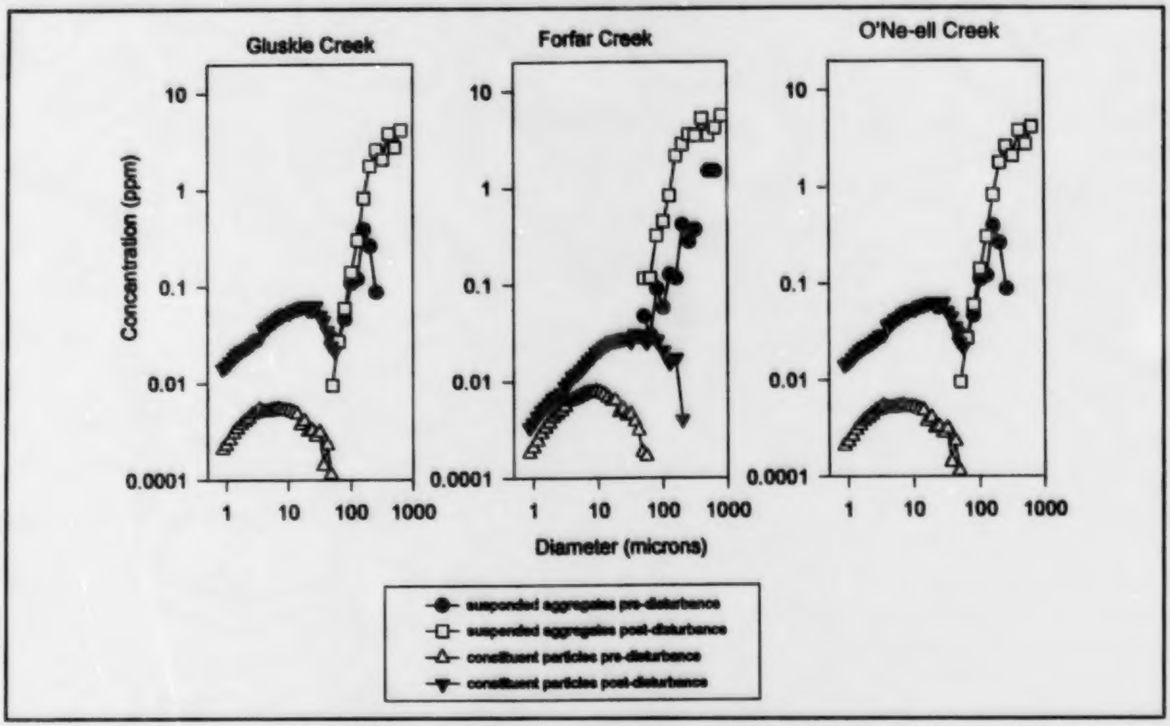

Figure 1. Grain size spectra for three creeks before and after gravel bed disturbances. The suspended aggregates represent the in situ particles that were sized from image analysis of underwater photographs. The constituent particle spectra were analyzed by Coulter counting and indicate the inorganic grain sizes that comprise the aggregates photographed in the water column.

fine-grained sediments are moving in the streams at low flows and following disturbances of the gravel beds. The observations of aggregates being transported in the stream and stored in the gravels (Figs. 1 and 2) indicates that settling rates of fines have been modified in these headwater streams. Therefore, the assumption of single-grained sediment transport for silt- and clay-sized particles is inappropriate for these systems.

Information about the origin and modification of the transported fines can be obtained from Figure 3. The mode size (8 μm) of the inorganic suspended sediment moving in O'Ne-ell Creek during low, undisturbed flow is identical to that of the gully material that represents the glaciolacustrine sediments underlying the lower portions of the three watersheds. The slopes of these two spectra below the mode size are different, indicating either that the riverine suspended sediment is not composed solely of the glaciolacustrine materials or that the gully sediments are not delivered directly to the stream as consolidated or aggregated particles. A recombination of the particles smaller than 8 μm has created the sediment spectra for the low-flow, pre-disturbance suspended sediments in O'Ne-ell Creek.

The spectrum representing the post-disturbance suspended sediment has a slope similar to that of the fine-grained sediments that are stored in the delta (Fig. 3). The lower portion (<20 μm) of these two spectra represents the constituent particles that constitute the aggregates that naturally settle out or are trapped in the gravel beds and delta sands. The constituent particle mode size (about 20 μm) of these samples is greater than that of the material moving during low flow (mode of 8 μm) implying an instream mixing of sediment source materials and/or a downstream reworking of the aggregated glaciolacustrine material.

Sediment size spectra of the delta cores indicate that fines are settling on the slope and prodelta regions and throughout the depths of all cores (Fig. 4). The composition (slope) of the fine-grained end of

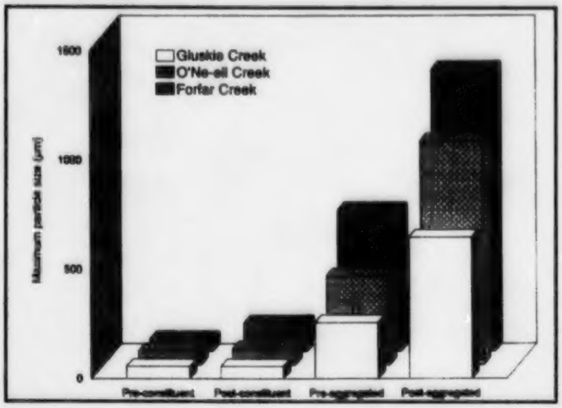

Figure 2. The maximum grain size of the suspended sediments sampled in each of the three creeks before and after gravel bed disturbance. The constituent particle and aggregate sizes are determined as mentioned in Figure 1.

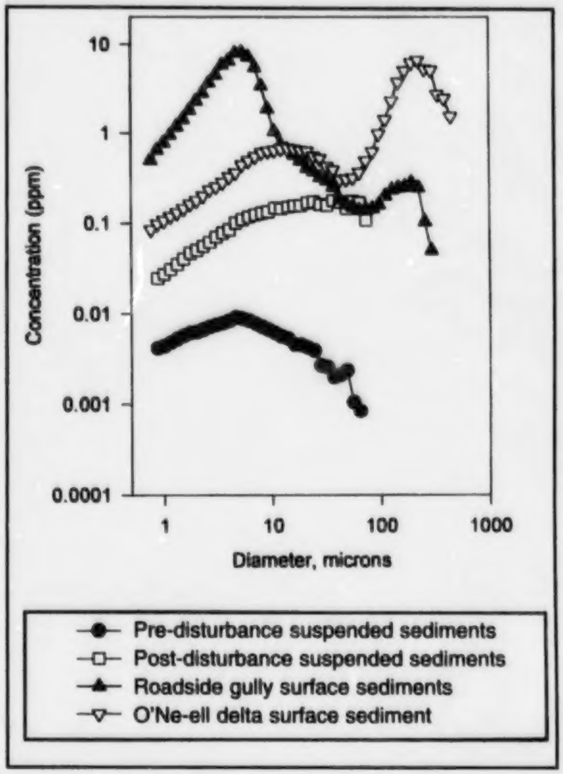

Figure 3. Inorganic sediment spectra comparing four types of samples from the O'Ne-ell Creek watershed.

the spectra is surprisingly similar over a range of energy environments (depths) and core depths (time of deposition). While the concentration of fines changes in the different energy environments (note position on the y-axis), the relative abundance of each particle size is maintained, indicating an aggregation or flocculation process is operating in these systems.

It should be reiterated that, while this baseline sample year (1994, pre-harvesting) indicates that the observed mass of fine material is small, relative to the sand-sized portion in these samples, the effects of forest harvesting are expected to result in changing conditions in the watershed (flow regime and sediment supply) that would increase fine-grained sediment delivery. Fine-grained retention in the gravel beds and delta is predicted, as we have observed that aggregation processes and storage of these sediments occurs over a range of energy conditions in these systems. The degree of change in the storage of fines and the effects on the quality of fish habitat will be studied in the next few years of the Stuart–Takla project as experimental cutting of the watershed is undertaken.

Conclusion

The presence of aggregates in headwater streams indicates that sediment transport in these systems is modified. This implies that the storage of fine-grained sediments (<63 μm) is accentuated in these environments, as the settling velocities of the aggrgates is greater than the settling speeds of the individual or constituent particles. This is verified by the observation that aggregates are being stored in both the gravel bed and in the sandy delta. Currently they do not comprise a large percentage of the mass of sediments, as fines are not being delivered to the channels in large quantities. It is unclear from this study if the aggregates are forming in the stream as a function of water chemistry and/or biological activity as noted in other freshwater and marine systems (Kranck et al. 1993; Droppo and Ongley 1994), or if some portion of them are being delivered to the stream as consolidated particles from the erosion of the glaciolacustrine deposits in the watershed.

The results of this preliminary, pre-harvesting study indicate that the quantity of fines stored in the channels and deltas is currently not significant relative to the mass of sands and gravels in the channels, and the small amount that is being stored is not reducing the quality of the fish habitat. An increased supply of fine-grained sediment is expected to result in increased gravel bed storage of fines and a concomitant decrease in habitat.

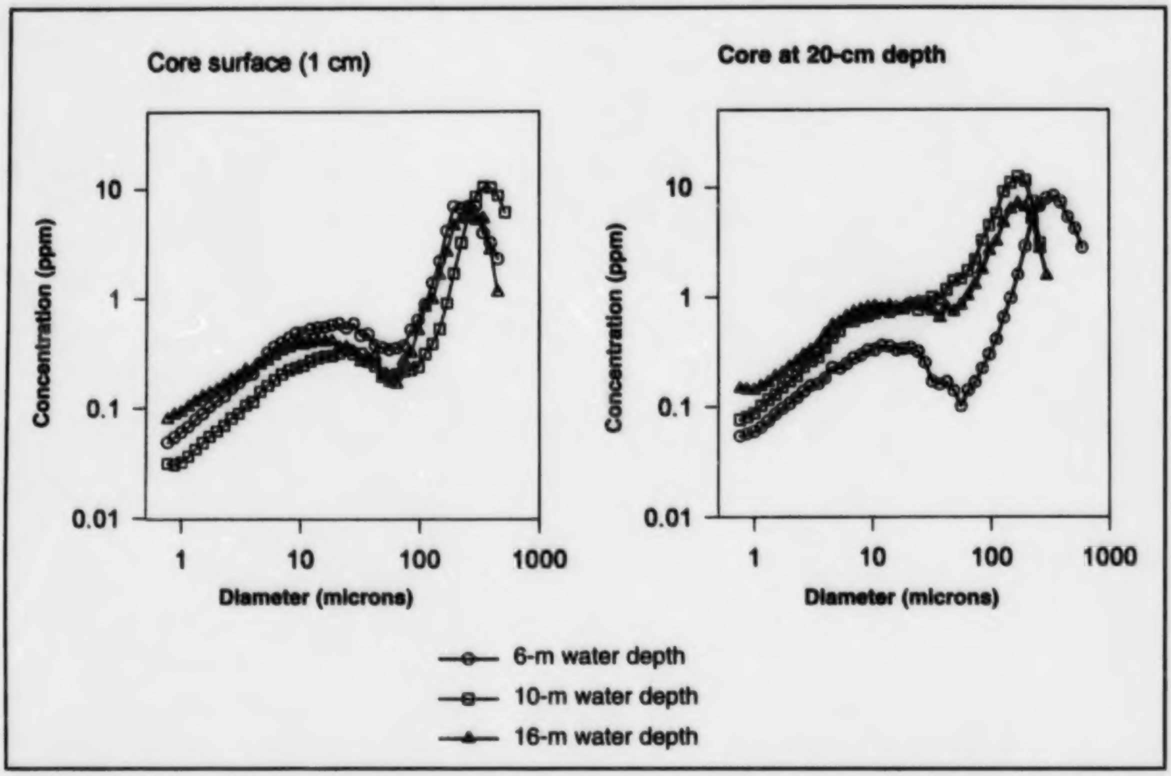

Figure 4. Inorganic sediment spectra of O'Ne-ell Creek delta samples at three water depths that represent the delta slope and prodelta sedimentary environments. Grain size composition from the core surface and at depth are presented to reflect recent and past settling conditions.

While the conditions influencing the process of aggregation is uncertain, fines are being transported as much larger aggregates, which allows them to be stored in the channel gravels and delta sands. The modified settling rates and resultant storage capacity are extremely important in the context of sediment routing, as they influence how quickly these systems can transport future deliveries of sediment loads that may occur as a function of disturbances in the watershed.

Acknowledgments

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Bedload Transport During Sockeye Redd Excavation and by Floods in the Stuart–Takla Experimental Watersheds, British Columbia

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Abstract

Movement of over 1400 bedload clasts was recorded from 1992 to 1995 in Forfar and O'Ne-ell (Kynoch) creeks, important salmon spawning streams of the upper Fraser River. Clasts were marked with neodymium magnets for relocation with a magnetometer. Over 8000 recoveries were made during the relocation surveys. Overall, at the five stations monitored, sockeye salmon (*Onchorhynchus nerka*) spawning activities resulted in 11% of the annual bedload transport, 4% on the higher gradient upstream stations, and 48% on the lower gradient downstream stations. Nearly all of the remainder of the bedload transport was by nival flood transport. In normal spawning years, the spawning of sockeye salmon reworks most of the streambed in the lower 3–4 km of the spawning streams, to an average depth of over 10 cm. Flood transport is more selective, with a smaller portion of the bed mobilized, but with longer transport distances and slightly greater depths. Sockeye salmon spawning results in the filling in of pools and the channel thalweg, and the widening of the effective channel. The intense bioturbation by sockeye salmon results in selective winnowing of fine sediments, and probably increases bedload permeability and improves egg to fry survival.

Introduction

The six species of Pacific salmon (*Onchorhynchus*) spend the majority of their lives in the ocean, returning to freshwater streams to spawn. When they reach the spawning streams, females defend select patches of gravel, and excavate composite spawning hollows called redds. After a pocket of eggs is laid and fertilized, another nest is excavated at the upstream end of the redd, and the gravel is deposited over the egg pocket downstream (Crisp and Carling 1989). Redd areas vary from 0.6 m^2 for the small pink salmon, *O. gorbuscha* (Hourston and MacKinnon 1957; Wells and MacNeil 1970) to over 5 m^2 for the largest species, chinook salmon, *O. tshawytscha* (Burner 1951). Sockeye salmon, *O. nerka*, of intermediate size, excavate redds averaging 1.8 m^2 (Burner 1951), with an average depth of 0.18 m (Scrivener 1994), for a volume of about 0.32 m^3. During spawning, at least this volume of gravel is transported downstream. Clearly, salmon might be significant agents of bedload transport in streams with high spawning populations.

I have collected a set of data to assess the significance of this transport process on Forfar and O'Ne-ell (Kynoch) creeks, tributaries of the Middle River, near the northern extreme of the Fraser River watershed,

Gottesfeld, A.S. 1998. Bedload transport during sockeye redd excavation and by floods in the Stuart–Takla experimental watersheds, British Columbia. Pages 249–258 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

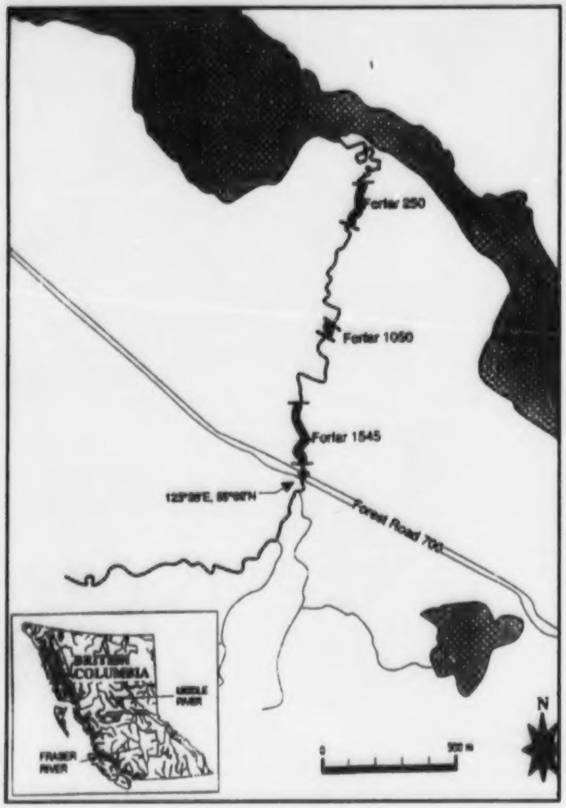

Figure 1. Study reaches Forfar Creek. Bedload movement was traced in the shaded zones. The inset map shows the location of the Middle River study areas in the upper Fraser River basin.

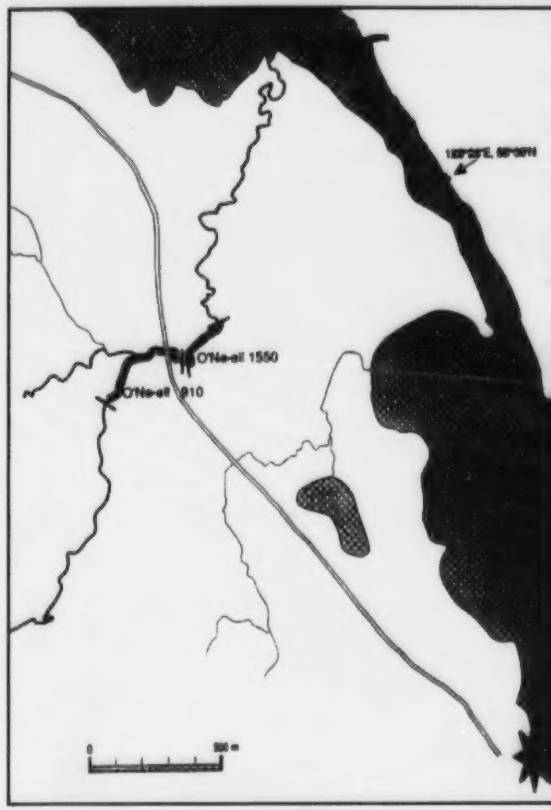

Figure 2. Study reaches O'Ne-ell Creek. Bedload movement was traced in the shaded zones.

British Columbia (Fig. 1). These creeks are important spawning streams of the early Stuart River sockeye salmon stock, which spawns from mid-July to mid-August. During the past 7 years, average annual spawning escapements were 11 100 (Forfar Creek) and 12 900 (O'Ne-ell Creek). These creeks were selected because they are undisturbed, are similar in size, and have few naturally magnetic rocks.

The drainages of Forfar and O'Ne-ell creeks have divides at approximately 1980 m, and mouths at slightly below 700 m. The annual precipitation is over 500 mm, with more then 40% falling as snow. These streams experience low flow during the winter, and are ice-covered from November to April. Both drainages are covered with boreal forest, which was last significantly disturbed by fire more than 50 years ago.

Methods

Three reaches on Forfar Creek (Fig. 1) and two reaches on O'Ne-ell Creek (Fig. 2) were selected for study along the lower portions of the streams where sockeye spawn. The study sites were named for the approximate distance above the mouth of the creek at which marked clasts were placed. Sockeye salmon spawning extends less than 1 km above the area shown in Figure 1, and several 100 m above the area shown in Figure 2.

Forfar and O'Ne-ell creeks are small nival-flow dominated streams, with widths of 8 to 15 m and peak discharges of 13 to 20 m^3/sec. Nival floods last several weeks and occur in May and early June, and may have several peak events in response to changing temperature, wind, and rainfall. The lower portions of Forfar and O'Ne-ell creeks show a downstream

decrease in slope from 0.02 to 0.005, and a slight decrease in average surface gravel size as they progress from the lower transport zone to the alluvial fan sections.

Gravel samples were collected from the upper 20 cm of riffles in reaches with relatively simple pool and riffle morphology. All clasts (pebbles and cobbles) larger than 5 cm were retained (n = 1379). Clast size, petrology, and roundness were recorded and a groove was cut with a diamond blade. Neodymium magnets and a numbered tag were embedded in epoxy which refilled the groove. Clasts were painted orange and replaced at their original position.

Clast movement was measured after each transport event: in April, following fall storms and possible winter and early spring-ice transport; in July, after nival-flood transport; and in September, after transport from bioturbation associated with sockeye salmon redd excavation. Clast placement areas were inspected several additional times during the year to detect movement during the fall and winter periods, and near the end of the snowmelt flood period. If clast movement was detected, an additional survey was undertaken. The clasts were relocated visually if they were on the surface, or with magnetometers if they were buried (Hassan et al. 1984; Hassan 1990; Hassan et al. 1991; Hassan and Church 1992; Schmidt and Ergenzinger 1992).

Over 8000 recoveries were surveyed with a Nikon D50 total station. Burial depths were estimated if less than 5 cm, or measured to the nearest centimetre, with a modified telescopic plumb pole. The survey instrument accuracy is within a few millimetres. Achieved survey accuracy was less, since it depended on where the survey plumb pole is placed on the clast surveyed, and surveying is difficult in rapidly moving water. Survey accuracy is estimated to be within 1 cm. Depths of burial were measured to the top of the clasts.

Except for some of the clasts at Forfar 1545 and O'Ne-ell 1910 in July 1992, recovered clasts were replaced along a band 20–40 cm wide at each study site. This simplified recovery, resulted in experimental runs which were highly similar, and avoided considerations of the variable likelihood of entrainment with different depths of burial (Hassan 1990; Hassan and Church 1994). At each replacement, clasts were pressed into the upper layer of the streambed. Clasts moved by more than one event were excluded from the analyses reported here.

Average distances of movement were calculated for the total set of clasts; depths of post-depositional burial were calculated for transported clasts only. Multi-site or multi-year averages for transport distance and burial are based on the averages of all observations, not the average of calculated site values.

Results

Due to the relatively small size and the shallowness of the creeks studied, high recovery rates were attained. Typically, after spring flood transport, with relatively high flows, recovery rates were 60–90% of the fall placement. In the late summer, following sockeye spawning bioturbation, recovery rates were over 90% of clasts in all cases, and we frequently recovered clasts missed in the spring. After 11 recoveries, we retained 59% of the original clasts.

Nival floods and spawning sockeye salmon together move most of the near surface streambed clasts each year (Fig. 3). For the 3 years sampled, the average proportion of clasts moved by nival floods was 60%, by sockeye salmon spawning 74%, and by fall and winter events 3%. In all 3 years sampled, the percentage of clasts moved by sockeye salmon exceeded the percentage transported by floods. In 2 of the 3 years, less than 50% of the bed material was moved by spring floods. Since the same clasts do not move on each occurrence, total annual rates of streambed disturbance are slightly higher than these values.

Although transport distances were as great as 340 m, most clasts were recovered within 20 m of their original position. The average annual transport distances for the 3 years monitored, based on 7936 observations at the five study sites, was 18.79 m (Table 1). The range in annual transport distance among the five sites is from 0.60 to 112.12 m (Table 2).

Nival floods moved streambed clasts average distances of 0.69 m to 70.88 m per year at the five study sites (Table 2) between 1992 and 1995. In the spring flood of 1992 the Forfar 1050 site had no nival movement; instead it experienced burial of 5 to 30 cm. In 1994 the O'Ne-ell 1550 site had no nival movement; it was buried 13.46 ± 0.84 cm.

An intense rain storm on June 28, 1993, near the end of the nival flow period, increased flows to 4.6 m^3/sec on Forfar Creek, resulting in some bedload transport. The transport distance of this event was recorded at three sites (Table 2.) The transport work performed by this event was included in the nival flood category.

Long transport distances in the spring of 1993 are related to a high peak discharges, which were not measured due to overtopping of the gauging sensors.

Table 1. Average annual bedload transport distances (m), Forfar and O'Ne-ell creeks, 1992–1995 (n = 7936)

Reaches	Nival flood	Sockeye spawning	Fall floods	Annual total
All	16.79	1.99	0.01	18.79
High stream power	31.24	1.34	0.03	32.61
Medium stream power	6.79	1.37	0.00	8.17
Low stream power	3.05	2.79	0.01	5.85

Table 2. Annual bedload transport distances (m) at the five study sites on Forfar and O'Ne-ell creeks, 1992–1995

Site transport	Nival flood transport	June flood transport	Sockeye spawning transport	Fall and winter transport	Annual total transport
Transport 1992–93					
Forfar 1545	9.61		0.32	0.004	9.94
Forfar 1050	0.00		0.60	0.002	0.60
Forfar 250	–		4.77	0.000	4.77[a]
O'Ne-ell 1910	35.11		0.33	0.024	35.5
O'Ne-ell 1550	4.88		0.18	0.022	5.08
Transport 1993–94					
Forfar 1545	28.82	4.35	1.86	0.003	35.03
Forfar 1050	10.24	0.30	2.93	0.000	13.47
Forfar 250	6.64	0.64	4.59	0.000	11.87
O'Ne-ell 1910	108.79		3.33	0.000	112.12
O'Ne-ell 1550	–		–	0.030	20.07[b]
Transport 1994–95					
Forfar 1545	25.78		1.20	0.039	27.02
Forfar 1050	4.36		1.41	0.003	5.77
Forfar 250	0.72		2.43	0.000	3.16
O'Ne-ell 1910	41.60		0.39	0.211	41.99
O'Ne-ell 1550	0.00		1.35	0.000	1.35
Transport 1992–95					
Forfar 1545 3 years	23.65	4.35	1.01	0.011	29.03
Forfar 1050 3 years	6.17	0.30	1.37	0.002	7.85
Forfar 250 3 years	3.77	0.64	3.99	0.000	8.40
O'Ne-ell 1910 3 years	70.88		2.15	0.049	73.08
O'Ne-ell 1550 2 years	0.69		0.61	0.025	1.33

[a] No data from nival transport.
[b] Only annual total obtained.

The 1994 flood was second in stage height, with peak discharges of 12.98 m^3/sec and 20.14 m^3/sec at Forfar and O'Ne-ell creeks, respectively.

Bioturbation by sockeye salmon moved clasts for shorter distances than nival floods (Tables 1 and 2 and Figs. 4 and 5), but transport occurred at all sites during each spawning period. In the course of redd excavation, clasts are ejected from the redd by pulses of water directed by the spawning salmon. The clasts then move along the bed for a distance dependent on the prevailing flow velocities. If another redd is excavated in previously moved material, the clasts move a second step downstream. Thus transport distance is dependent on the density of spawners and the stream stage. Most spawning occurs during low flows, but occasionally, as in 1993, intense rain increases the water flow to moderate values for 1 to several days.

Fall flood and winter ice-related movement were not important and affected less than 3% of the clasts.

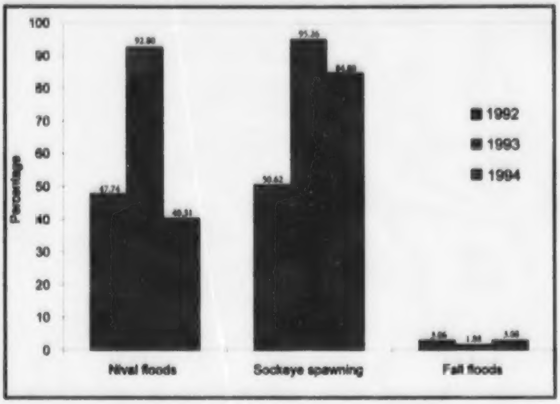

Figure 3. Percentage of clasts moved at five reaches on Forfar and O'Ne-ell creeks, 1992 (n = 1223), 1993 (n = 914), and 1994 (n = 813).

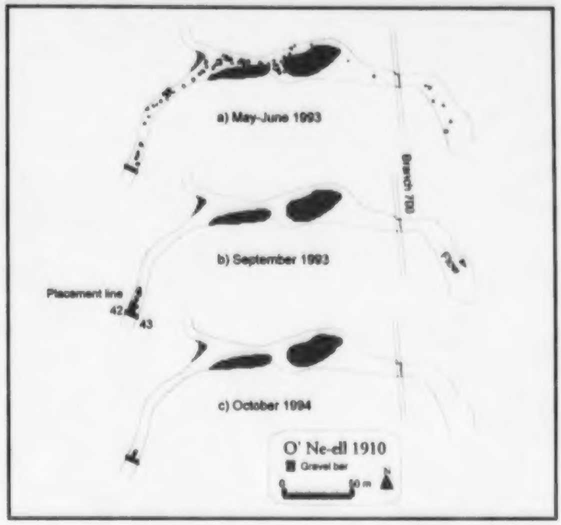

Figure 4. Bedload transport at O'Ne-ell 1910, a high stream power site. Only the locations of transported clasts (black dots) are shown; other clasts remain at the placement line.

a) Nival flood transport during the nival flood of May and June 1993.

b) Bedload transport during sockeye redd excavation.

c) Bedload transport during a fall storm, October 25–26, 1994.

Transport distances of moved clasts were short (Tables 1 and 2). Probably most of the fall and winter movement took place before the streams froze. The October 25–26, 1994, storm resulted in more transport of clasts than the total transport of either of the previous two winters (Fig. 4). No further transport occurred during the remainder of the winter of 1994–95, which was relatively mild.

Figures 4 and 5 show examples of bedload transport during nival floods, sockeye salmon redd excavation, and fall floods. O'Ne-ell 1910 (Fig. 4) has a relatively steep slope (0.010) and had the longest nival transport distances. The nival flood of May and June 1993, when peak discharge exceeded 12.65 m^3/sec, was the highest discharge and longest distance of bedload transport observed (Fig. 4a). Air photo examinations of the channel history of the past 40 years suggest that this was a flood with a recurrence interval of less than 10 years, and probably closer to 5 years. All clasts moved in this event. The average transport distance was 108.79 m; the maximum transport distance was 343.56 m. The uneven pattern of deposition over the first 200 m reflects the pool–riffle spacing, with deposition occurring preferentially on riffles along the thalweg.

Bedload transport accompanying sockeye salmon redd excavation in late July to mid-August 1993 (Fig. 4b) occurred during estimated discharges between 0.5 and 2.2 m^3/sec. Mobilized clasts moved 3.61 m. The average distance of transport distance was 3.33 m with 92% of the clasts moved. The maximum transport distance was 19.10 m.

During the intense fall rain-on-snow event of October 25–26, 1994 (Fig. 4c) 15% of the marked clasts moved. Peak discharge was 2.50 m^3/sec. The average distance of transport was 0.211 m. The average transport distance of the displaced clasts was 1.38 m with a maximum transport distance of 6.23 m. This was the maximum distance of fall or winter transport observed at any site during the 3 years of the study.

Forfar 250 (Fig. 5) is a low gradient site (0.005) with relatively high sockeye spawning density (Tschaplinski 1994) and typically short transport distances. The peak discharge of the May to June nival flood in 1994 was approximately 3.95 m^3/sec on 7 June. This resulted in the movement of 36% of the marked clasts (Fig. 5a). The maximum distance of movement was 16.70 m with an average transport distance of moved clasts of 2.00 m. The average distance of movement of all clasts was 0.72 m.

In August 1993 (Fig. 5b) transport during sockeye redd excavation moved 99% of the marked clasts. Water discharge varied between 0.46 and 2.10 m^3/sec. Transported clasts moved 4.65 m; the

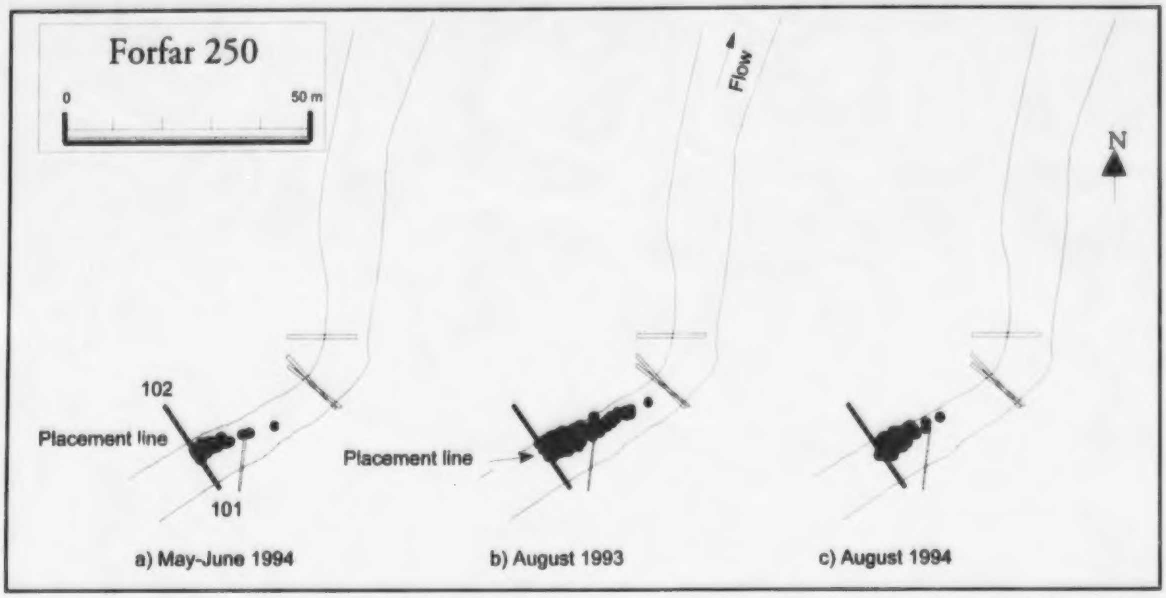

Figure 5. Bedload transport at Forfar 250, a low stream power site. Only the locations of transported clasts (black dots) are shown; other clasts remain at the placement line.

a) Nival flood transport during May and June 1994.

b) Bedload transport during sockeye redd excavation, August 1993.

c) Bedload transport during sockeye redd excavation, August 1994.

average distance of transport of all clasts was 4.59 m. The maximum transport distance was 23.10 m. In August 1994 (Fig. 5c) transport during sockeye redd excavation also moved 99% of the marked clasts, while the water discharge varied between 0.43 and 1.03 m^3/sec. The transported clasts moved an average distance of 2.45 m; the average distance of transport of all clasts was 2.43 m. The maximum transport distance was 14.09 m.

Most transported clasts were recovered at or near the surface, although some deep burial took place (Fig. 6). During the large flood event in late May 1993, local scour and subsequent filling resulted in a maximum burial depth of 77 cm. The filling in of pools during the low flow conditions that occur during sockeye spawning resulted in burial as deep as 45 cm.

Bioturbation accompanying redd excavation in the summer of 1993 uncovered and moved clasts buried >30 cm at Forfar 1050 during the previous nival flood. Further downstream at Forfar 250 many clasts were recovered lying on the surface of the earlier infilled thalweg below the placement site. Apparently redd excavation removed many of the clasts early in the spawning interval, and subsequent

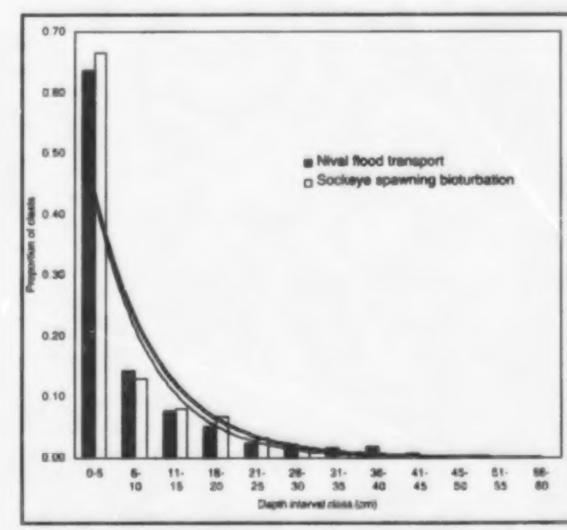

Figure 6. Depth of burial of transported clasts in Forfar and O'Ne-eil creeks, 1993 and 1994. The exponential curve for the nival flood transport ($y = 0.821e^{-0.636}$) is the thick line; the sockeye spawning induced bioturbation ($y = 0.851e^{-0.697}$) is the thin line. The two distributions show similar negative exponential distributions.

Table 3. Depth of burial of clasts after transport (cm), 1992–94. The average depth is computed from the total set of transported clasts. Clasts at O'Ne-ell 1550 in 1994, and Forfar 1050 in 1992 were buried in place during the nival floods. The c-axis measurement is the average thickness of the clasts in each set, and is the approximate thickness to add for scour depth.

Site	Nival floods	n	Sockeye spawning	n	Clast c-axis
1992					
Forfar 250			7.41 ± 0.95	264	3.78
Forfar 1050	0.00	0	1.28 ± 0.33	155	6.53
Forfar 1545	2.89 ± 2.45	38	1.35 ± 0.44	66	4.64
O'Ne-ell 1550	6.70 ± 5.10	10	14.26 ± 4.81	35	4.24
O'Ne-ell 1910	1.73 ± 1.02	37	0.63 ± 0.49	27	4.88
1993					
Forfar 250	14.12 ± 1.95	178	14.30 ± 1.21	237	3.78
Forfar 1050	7.93 ± 1.50	205	5.32 ± 1.07	228	6.53
Forfar 1545	4.51 ± 1.55	129	3.44 ± 0.77	141	4.64
O'Ne-ell 1910	2.98 ± 1.60	43	3.98 ± 0.88	98	4.88
1994					
Forfar 250	0.45 ± 0.22	84	6.76 ± 0.91	230	3.78
Forfar 1050	9.41 ± 1.86	96	1.94 ± 0.58	116	6.53
Forfar 1545	1.66 ± 0.78	90	0.77 ± 0.32	97	4.64
O'Ne-ell 1550	0.00	0	2.06 ± 0.48	159	4.24
O'Ne-ell 1910	2.89 ± 1.51	38	3.72 ± 1.02	46	4.88
Average depth	6.63 ± 0.65	948	5.58 ± 0.35	1899	4.83

bioturbation buried them as much as 40 cm (mean = 19.5 cm).

The average depth of burial of clasts after nival flood transport and after sockeye salmon spawning transport are similar (6.63 ± 0.65 and 5.58 ± 0.35 cm), although the depth of burial varies with each transport event (Table 3). In general, burial depths were greater in 1993 than in 1994 because of greater volumes of material in transport.

Little scour and deposition accompanied the small magnitude transport events. All clasts were found at the surface after transport during the fall or winter seasons. The average depth of burial of clasts at the three study reaches of Forfar Creek following the minor flood of June 1993 was 0.3 cm (n = 252).

The proportion of bedload transport due to nival flood transport, sockeye salmon bioturbation, and fall and winter events was calculated by comparing the total transport distances of all marked clasts. This calculation assumes that the five strips of marked clasts adequately represent the streambed. For this analysis, the experimental reaches are grouped by stream power, the product of specific weight of water, discharge, and slope (Bagnold 1977). I have assumed that sediment concentrations are not high enough to significantly affect the density of water and therefore I used the value 9.81 for stream power calculations. The stream power values (Table 4) permit the separation of the reaches into three groups. The high power reaches are Forfar 1545 and O'Ne-ell 1910, the medium power reach is Forfar 1050, and the low power reaches are Forfar 250 and O'Ne-ell 1550.

Table 4. Stream power of Forfar and O'Ne-ell creek study reaches

Reach	Slope	Peak flow	Stream power
Forfar 250	0.005	12.98	0.637
Forfar 1050	0.009	12.98	1.146
Forfar 1545	0.017	12.98	2.165
O'Ne-ell 1550	0.005	20.14	0.988
O'Ne-ell 1910	0.010	20.14	1.976

Bedload transport work declines with decreasing stream power (Table 5). Nival flood transport was dominant at all sites, but only marginally so at the lowest gradient reaches. As stream power declines, the proportion of the transport accomplished by sockeye spawning bioturbation increases from 4.09% of the transport work at the high gradient sites to 47.54% of the transport work at the low gradient sites. At all sites, fall storm transport and ice transport were insignificant.

Table 5. Bedload transport work proportions, Forfar and O'Ne-ell creeks 1992–1995. Transport work is calculated as the proportion of total travel length.

Reaches	Nival flood (%)	Sockeye spawning (%)	Fall flood (%)
All 89.33	10.60	0.07	
High stream power	96.81	4.09	0.10
Medium stream power	83.16	16.82	0.02
Low stream power	52.26	47.61	0.13

Discussion

There are two important processes of bedload movement in the sockeye salmon spawning creeks of the Stuart–Takla area. Nival flood bedload transport is dominant, as it is in most boreal forest streams. However, bedload transport effectiveness is variable, depending on the nature of the snowmelt, and may not affect large areas of the streambed every year. Nival flood transport, when it occurs, results in relatively long distances of transport. Transport related to sockeye spawning is second in importance, occurs every year, and affects more of the streambed than flood actions. The amount of bedload transport is directly related to the spawner escapement and the water discharge at the time of spawning.

Fall and winter events result in little bedload movement, and transport distances are short. Although some transport related to winter ice is likely, the bulk of the transport probably occurs during fall frontal storm precipitation augmented by snowmelt (Harr 1981).

In the streams studied, as the slope of the stream decreases the flood transport distances decrease, but the sockeye salmon spawning induced transport distances increase, presumably because of more favorable spawning habitat in the lower gradient reaches.

As clasts are transported downstream they may come to rest on the newly scoured streambed or within the newly deposited layer. If the marked clasts are moved early in the transport event, they may be buried where they come to rest on the preexisting or newly eroded surface. Clasts moved at the end of the transport event are more likely to end up at the surface.

In streams which are neither aggrading nor degrading, a category which includes the Middle River tributaries, average scour and average fill may be assumed to be the same over a sufficiently long reach. Hassan (1990) measured burial to the base of the clasts and found that the average depth of burial of transported clasts approximates the thickness of streambed scour and fill. The average depositional depths reported here underestimate scour depth, since they were measured to the upper surface of the buried clasts, which average 4.8 cm thick. Average values of the c-axis of each set of marked clasts are reported in Table 3 and can be added to the average depth of burial of clasts to approximate scour and fill depths.

According to Scrivener (1994), the average depth of sockeye salmon egg clusters is 18 cm. If we assume that the sockeye salmon egg pockets are constructed at the deep end of the redd, then an average excavation depth (burial depth of transported clasts + c-axis length) of 10.4 cm seems reasonable.

In Forfar and O'Ne-ell creeks most clasts are deposited within the surface layer or at shallow depths. The distribution of clast burial depths (Fig. 6) suggests that streambed movement is complex and cannot be modeled as a single mobile layer (Hassan and Church 1994). A more appropriate model would show extensive mobilization of an upper layer a single cobble thick, and sporadic scour and fill with exponentially decreasing probability of deep scour.

The depths of burial after nival flood transport demonstrate this negative exponential decline with depth. This type of distribution is to be expected after single movements by floods (Hassan and Church 1994). Since the depth measurements plotted in Figure 6 are based on clasts replaced in their original position before each transport event, this negative exponential distribution may be expected. Surprisingly the distribution of depths after sockeye spawning related bioturbation shows a similar negative exponential distribution despite important differences in the processes.

There appears to be a tendency for the depths of burial following bioturbation by sockeye to be similar to those of the preceding nival flood transport burial ($r^2 = 0.36$). This might result from interactions between these two processes. The freshly deposited gravels from the spring floods are easily excavated

and may provide preferred sites for salmon spawning a month or two later. Similarly the spawning of salmon in the late summer results in loose deposits of gravel in sites, such as pools, which are easily excavated in the higher spring nival flows about 9 months later. Thus the efficiency of bedload transport may be raised for both processes, and there may be a tendency for the volume in transport to be relatively constant from year to year.

The spawning transport estimates for 1992 to 1995 are probably lower than the long-term average. In years with strong salmon runs, commercial fishing may take over 70% of the returning salmon (Langer et al. 1992). In 1992 and 1994, heavy fishing pressure and poor in-river migration conditions resulted in unusually low numbers of returning sockeye salmon of the early Stuart River stock. Nevertheless, the extent of sockeye salmon related transport is impressive.

Since salmon spawning on Forfar and O'Ne-ell creeks occurs at low and medium stages, when velocities are too low for normal bedload transport, only sand-sized and finer sediments have long transport distances. This results in export of fine sediments to Middle River during sockeye salmon spawning (Cheong et al. 1995). Bioturbation from salmon spawning coarsens the spawning gravels (Kondolf et al. 1993) and produces a loose skeletal framework in the redd gravels. This gravel structure presumably enhances egg survival by increasing the permeability of the spawning gravels, and facilitates intragravel passage of alevins (Reiser and Bjornn 1979; Huntington 1985; Everest et al. 1987; Chapman 1988; Scrivener 1994).

Bioturbation by sockeye salmon may also alter the bed forms of the spawning creeks (Huntington 1985; Tutty 1986) and their pool–riffle sequences. In general, transport from sockeye salmon spawning appears to result in infilling of the thalweg, widening of the wetted portion of the channel, extension of riffles, and filling in of pools. We are currently collecting serial surveys of channel reaches to quantify these observations.

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Windthrow Risk Assessment and Management in Riparian Management Areas in Coastal British Columbia

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Abstract

Windthrow damage to riparian management areas is a concern in many areas of coastal British Columbia (B.C.). Under the recently enacted *Forest Practices Code of British Columbia Act,* lakes and streams must be classified and management areas of specific widths are required. For example, fish-bearing creeks from 1.5 to 5 m wide require 20-m wide riparian reserves plus 20-m wide management zones to buffer the reserve. In response to these new requirements a variety of management options are being developed in coastal B.C. Windthrow risk assessment frameworks have been developed. Feathering of management zones is being used in multi-story stands with moderate windthrow risk. In areas determined to have high windthrow risk, topping, and pruning techniques are being tested. Recent innovations in helicopter suspended trimmers enable removal of the top branches from trees. These safe, inexpensive techniques for directly treating reserve zone trees have won support from licensee and agency representatives in the locations where they have been applied. Research into the long-term effectiveness of topping and feathering treatments is underway.

Mitchell, S.J. 1998. Windthrow risk assessment and management in riparian management areas in coastal British Columbia. Pages 259–262 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

Maintaining residual streamside vegetation is desirable to provide shade, a supply of leaves and insect drop, visual cover, bank stability, large woody debris (LWD), and reduction of logging debris and sediment entry into streams (Toews and Brownlee 1981). Streamside reserves also provide habitat for bird and wildlife species and improve the visual appearance and recreational value of managed forest land. Wind damage to partially or fully forested streamside strips occurs frequently in coastal B.C. and is commonly cited as a management concern by forest managers (Mitchell 1995). Post-harvest windthrow can lead to partial or total loss of shade, entry of large quantities of branches into the stream channel, and exposure of mineral soil (Moore 1977). Windthrow has also been implicated in slope and gully instability (Lutz 1960; Swanston 1967). Unsalvaged windthrow can provide rearing habitat for ambrosia beetles (*Trypodendron* spp. and *Gnathotrichus* spp.), which spread to and degrade felled timber (McLean 1985).

The impacts of windthrow on fish habitat are not always negative. Windthrow is a natural event in coastal stands and is a major source of LWD inputs into streams (Steinblums et al. 1984). In a study of 30 1- to 6-year-old streamside buffers in Oregon, Andrus and Froehlich (1992) found that only trees growing within or beside the stream channel were likely to become sediment sources, and fewer than 20% of the trees that fell actually had their boles or tops within reach of high water. In determining appropriate management responses for a given site, the expected level of windthrow should be assessed, the potential beneficial and deleterious effects of windthrow should be considered, and the acceptable level of windthrow determined.

Forest Practices Code

Under the recently enacted Forest Practices Code (British Columbia Ministry of Forests 1995) administered by the B.C. Ministry of Forests (BCMOF) and B.C. Ministry of Environment (BCMOE), streams are classified based on use by fish and channel width. Reserve and management zone widths vary with stream class (Table 1). Where reserve zones are required, it is intended that they be disturbed as little as possible by management activities. The management zone buffers the reserve zone and together these two zones form the riparian management area. The best management approaches for each stream class are summarized in the Riparian Management Area Guidebook (British Columbia Ministry of Forests and British Columbia Ministry of Environment 1995a). Where modifications to the management zone or reserve zone widths set out by legislation are proposed due to concerns about windfirmness, they must be supported by a windthrow risk assessment. The Gully Assessment Procedure Guidebook (British Columbia Ministry of Forests and British Columbia Ministry of Environment 1995b) contains a summary of pre-logging management strategies for various classes of forested gullies. On gullies with moderate or high downstream impact potential and high or moderate water transport, fan destabilization, or debris flow initiation potential, suggested management strategies include leaving the gully unlogged and buffered. The guidebook underlines the need for windfirm boundaries in cases where treed gully reserves are left.

Assessing Windthrow Risk

There is, to date, no officially required method for windthrow risk assessment in B.C. The Windthrow Handbook for B.C. Forests (Stathers et al. 1994)

Table 1. Stream classes, riparian reserve zone, and riparian management zone widths

Stream width (m)	Fish stream	Stream class	Riparian reserve zone width (m)	Riparian management zone width (m)	Riparian management area width (m)
>20	Yes	S1	50	20	70
5–20	Yes	S2	30	20	50
1.5–5	Yes	S3	20	20	40
<1.5	Yes	S4	0	30	30
>3	No	S5	0	30	30
<3	No	S6	0	20	20

summarizes windthrow concepts and management strategies and lays out a checklist of windthrow risk indicators. A simple framework for classifying biophysical factors (soil, stand, and topography) and management factors (cutblock shape, size, and orientation) to assess relative windthrow risk is presented in Mitchell (1998). In this framework, windthrow risk is considered to be the intensity of damage caused by endemic peak winds. This framework has been adopted for the purpose of instructing practitioners, and is being tested in current studies.

Endemic peak winds are peak winds which recur in a given location every 2–3 years or less. In coastal B.C. the two major sources of peak winds are Pacific low pressure systems which produce high southeast and northwest winds along the coast, and arctic outbreaks which produce high easterly winds in the mainland inlets and across Vancouver Island. Both of these winds are strongest in winter months (Environment Canada 1992). Key indicators of high windthrow risk for riparian management areas include shallow rooting, poor drainage, high pre-harvest stand density, and high topographic exposure due to funneling or speed-up of winds. Windthrow is more likely where leave strips are perpendicular to wind flow (Moore 1977).

The Riparian Management Area Guidebook contains strategies for improving the windfirmness of riparian management areas, including placement of boundaries in more windfirm locations and avoidance of boundary indentations or projections. Where moderate or high windthrow risk is determined, managers can vary from the best treatment options in the guidebook. Alternative treatments currently being used in coastal B.C. include: full retention of both management and reserve zone trees; partial retention or feathering of management zone trees to buffer reserve zone trees; removal of trees from the management zone combined with topping or top-pruning of reserve zone trees; and complete removal of overstory from both the management and reserve zones. Complete removal of the overstory may be accompanied by understory or safe-snag retention. The choice between alternative treatments is made in consideration of potential impacts with and without treatment, and in consultation with BCMOF and BCMOE staff.

Edge Feathering

The objective of edge feathering is to increase the permeability of upwind stand boundaries to wind so that wind speed at canopy height is reduced. Edges recently exposed by harvesting behave in one of three ways, they remain undamaged, they are partially damaged within a tree length or two of the edge (natural feathering), or they are heavily damaged for several tree lengths (progressive damage). Pre-feathering management zone edges during harvesting should be considered where there is evidence that harvesting boundaries with similar characteristics undergo natural feathering. The objective in feathering is to remove the trees with the lowest windfirmness. Trees with asymmetric or stilt roots, and trees which are rooted on unstable substrates like old logs or rootwads are typically less windfirm. Veterans or trees with broken tops are often more windfirm (Stathers et al. 1994). Feathering is unlikely to be effective where progressive windthrow has occurred on adjacent harvesting boundaries with similar characteristics. In these situations removal of management zone trees and topping or top-pruning of reserve zone trees should be considered. In a study of edge feathering and edge serration in an old growth stand on northern Vancouver Island, Gillies (1997a) concluded that while both treatments are operationally feasible, feathering is more costly because of the need for precise directional falling. A cooperative project between the University of B.C., BCMOF Vancouver Region, and Western Forest Products Limited (Port McNeill, B.C.) is underway to measure the properties of clearcut edges in the field and to model the behavior of feathered edges in a wind tunnel.

Topping and Top-pruning

Tree topping and branch pruning techniques have recently been extended to forestry practice in coastal B.C. in an attempt to preserve riparian and gully reserve trees in high hazard areas. During 1995, the BCMOF Vancouver Forest Region, Western Forest Products Limited, and the Forest Engineering Institute of Canada (FERIC, Vancouver, B.C.) cooperated in a study of manual and aerial topping and pruning techniques in a streamside reserve on northern Vancouver Island (Boswell 1997). Subsequent experimentation has lead to the development of several inexpensive helicopter pruning techniques which have been used operationally on a number of riparian and gully reserves (Gillies 1997b, c). With these techniques the branches from the upper 30% of the crown of dominants and co-dominants within the reserve zone are removed using either shearing bars or saws suspended below the helicopter. In second growth stands with branch base diameters less than 10 cm, costs of approximately $35–45 per treated tree are typical.

Conclusion

The Forest Practices Code requirements for enhanced riparian and gully management have lead to an increased need for windthrow assessment and management in B.C. In response to recurrent losses of riparian reserves, a variety of management techniques are being applied. A number of cooperative research projects are being initiated under Forest Renewal B.C. funding to investigate the long-term effectiveness of these techniques.

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A Private Land Forest Inventory for Central Alberta and Some Possible Uses

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Abstract

Although private forests in Alberta are under increasing harvest pressure, no basic inventory exists to improve management of this resource. An inventory of private lands in central Alberta, nearly complete, promises to be a useful resource. This inventory covers an area roughly from Calgary to Edmonton and east to the Saskatchewan border and incorporates both analysis of Landsat imagery and interpretation of air photos. This inventory will provide the basis for improved private forest management and could also include land use and watershed planning, rural economic development, and extension and awareness programs.

Introduction

While there is little quantitative information about privately owned forests in Alberta, which might contribute to their management, there is increasing demand on the resource. An often-quoted but difficult-to-reference estimate, states there are 1.2 million ha of forested private land in Alberta. *The State of Canada's Forests* (Canadian Forest Service 1995) indicates that 4% of Alberta's total forest area of 38.2 million ha is privately owned. Neither estimate includes any indications of species distribution, age, height, volume, or location of timber specific to private forests.

Over a very short period of time, private forests have become economically significant in Alberta. During late 1994 and early 1995, lumber companies from British Columbia offered stumpage rates much higher than previously available to Albertans, making forest liquidation tempting to landowners (Ezra Consulting 1996).

A multi-year summary of timber harvested from private lands provided by Ezra Consulting (1996) indicates that 2.87 million m^3 of private land timber was harvested in 1994. About one-half of this total volume was exported to B.C. The total was nearly a three-fold increase from the 1992–93 harvest levels reported by Alberta Environmental Protection (AEP) (1994). To help put 2.87 million m^3, and the export volume, into perspective, it was commonly reported in Alberta newspapers that up to three hundred loaded logging trucks, each carrying 30–40 m^3 of timber, were leaving Alberta for B.C. on a daily basis. Even if this estimate is exaggerated, this unprecedented timber removal from private lands raised the questions of sustainability and environmental impact. Would this be a one-time harvest, or could private forests provide for a sustainable woodlot industry in Alberta?

A number of factors contributed to relatively dramatic increases in timber values paid to Alberta's timber owners. Two of the most significant causes are a

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significant decrease in the amount of timber available for harvest in major producing areas such as B.C. and the Pacific Northwest of the U.S.A., due to environmental considerations, and a strong demand and price for both softwood and hardwood.

Timber liquidation and land conversion created concerns for many Albertans as well as provincial and municipal governments. Concerns included loss of wildlife habitat and aesthetic values, loss of local job opportunities, negative impacts on water quality, improper land use practices, and damage to roads.

Given the expected long-term increase in world-wide fiber demand (Canadian Forest Service 1995), and the possible role private forests might play in meeting that demand, the question of whether land should be managed solely for agriculture or include management for woodlots may be an important one. Unfortunately, the concept of a private woodlot industry is new to Alberta so this question is seldom asked.

For landowners, there was often little time or information available to allow logging decisions to be made on an informed basis. For municipalities containing large amounts of private timber, there was limited or no existing data on local timber and other related resources to help guide their response. Inventory data could be a significant aid to targeting extension efforts, forming logging guidelines, and promoting the long-term perspective on economic returns from private forests. Similarly, senior governments also lacked basic data necessary to develop policies quickly and to tackle the broader issues related to the public good associated with use of privately owned forest land.

Objectives

Given the harvest pressure on private forests during 1994 and the lack of basic information about them, there was a demonstrated need for a basic inventory through which a variety of needs could be met. The goal of the project was to create an inventory of private forests in central Alberta. The inventory would provide a database to be used by resource managers and agencies to meet at least five objectives: 1) be suitable for municipal scale planning; 2) be relatively inexpensive; 3) provide coverage for a large area; 4) be completed in a short period of time; and 5) be readily updated.

In addition to meeting these objectives, it was hoped that the forest inventory, plus other complementary natural resource inventories, would be used by municipal governments, the Woodlot Association of Alberta, and resource agencies to assist in the development of an agro-forestry industry. Agro-forestry could have the potential to diversify the rural economy through: 1) the direct value of timber harvest: 2) value-added processing of timber products; and 3) the sale of special forest products such as craft products, wild edibles, and essential oils.

Methods

The project was recognized to be beyond the scope of any one agency so a partnership consisting of five members was formed. The partners include the Canadian Forest Service, Prairie Farm Rehabilitation Administration, Resource Data Division of AEP, two branches of Alberta Agriculture Food and Rural Development, and Alberta Pacific Forest Industries.

The inventory covers an area running north from about Calgary to Edmonton and then east to the Saskatchewan border (Fig. 1) The study covers an area of about 75 000 km^2 and includes portions of the Foothills, Boreal Mixedwood, and Central Parkland natural regions (Alberta Environmental Protection n.d.). It is expected that a similar inventory will be completed for the northeast portion of the province within the next few years.

Creating the maps consisted of two main steps. This first step was classification of Landsat TM data and the second was air photo interpretation. Supervised classification was used to differentiate forested areas from other cover types and provide an outline of all forested parcels (Pearson Timberline Forestry Consultants 1996).

Base map information, along with outlines of forested polygons, was drawn onto transparent Vernatec paper plots, in order to reference air photos of 1:30 000 and up to 1:60 000 scale. The most recent air photos available were then used to determine forest type information for each polygon. Attributes were determined from air photos and attached to each polygon in digital format. Initial polygons were broken into smaller parcels if air photo interpretation determined that polygons consisted of more than one map type. In order to keep the number of polygons with complete information manageable, any polygon less than [illegible] ha in size was filtered out prior to the air photo interpretation stage. In areas where detailed forest information was available through Alberta Vegetation Inventory (AVI) (Alberta Environmental Protection 1991, 1993), this data was generalized using a program developed by AEP to match the project legend and scale. This provided continuous coverage for the project area at one scale.

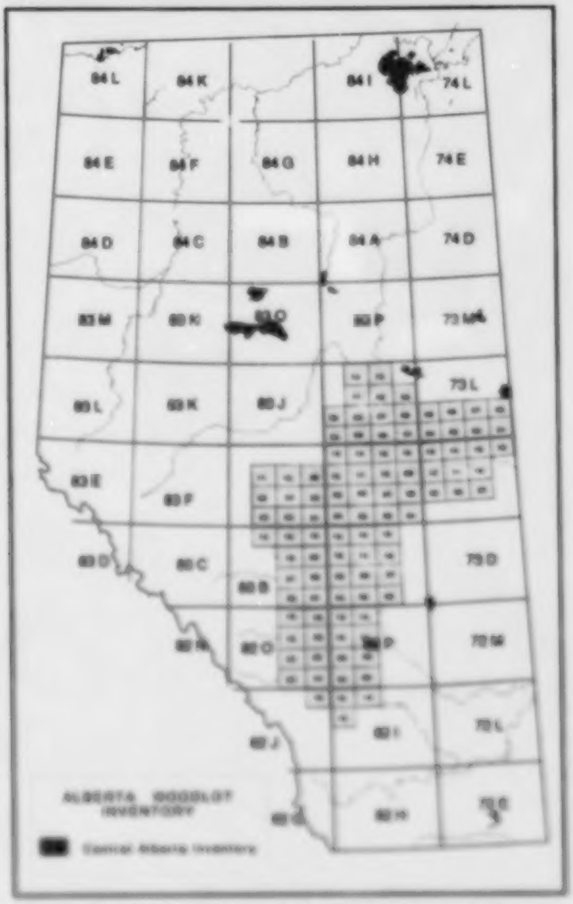

Figure 1. The private land forest inventory study area.

The work was initiated in the winter of 1995 and scheduled for completion by end of the summer of 1996. Analysis was based on September 1994 satellite imagery. Satellite images from 1995 have also been obtained to test the feasibility of determining forest parcel depletion.

The criteria used to classify forest blocks included three height classes, two density classes, two moisture types, and four stand composition indicators. The height classes used were greater than 15 m, 10-15 m, and less than 10 m. Stand densities used were 6-30% crown closure and greater than 30% crown closure. Moisture classes used were upland and wetland. Stand compositions were deciduous (>80% deciduous), dominantly deciduous (50-80% deciduous), coniferous (>80% coniferous) and dominantly coniferous (50-80% coniferous). The maps also indicate whether land is privately owned or crown land.

Definitions of polygons and their attributes were all stored in a digital file with a base map overlay. The base map overlay (1:20 000 scale) includes a township fabric, water bodies, roads, and other civil information.

The final product will be available both as colored paper maps, at 1:50 000, and as digital polygon information files.

Results

The process of using a combination of interpreted satellite imagery to delineate forested areas and air photo interpretation to describe the forest polygons was effective where forest polygons were relatively small. When forested areas became large and contained several forest types, it was equally as effective to use air photo interpretation only. In mostly forested areas the cost of interpretation climbed steeply, while in areas of scattered forest the two-step system was very cost-effective. For the 97 map sheets classified, the average cost for each 1:50 000 map sheet was about $2500.

Testing of the process' ability to be used for forest cover updating has been undertaken; 1995 images were compared to the 1994 base map and forest depletions were successfully determined. The process was unable, however, to determine selective forest removal.

When AVI data was generalized to match the legend created for the private land forest inventory project, the process appears to have produced good boundary matches and satisfactory reclassification.

Discussion

If there is enough private forests remaining in Alberta, a sustainable woodlot industry could develop and contribute significantly to economic opportunities and to rural diversification. A completed inventory will help identify the opportunities and issues associated with this resource. It could form the basis of a coordinated and informed decision-making process that will allow private forests to play a positive role in sustainable rural communities.

In eastern Canada (Ross 1996) and much of the United States, private forests are the major source of [illegible] and fiber for the forest industry. In Alberta, [illegible] forest companies are allocated 20% of their [illegible] through forest management agreements on crown land and have to compete for the remaining [illegible] through a bid process for crown timber or by [illegible] to private land owners. [illegible] land owners more valuable as competition for

timber on crown land increases. Indications of this trend are born out by increasing interest in the purchase and management of private land timber by industry (K. Glover, 1996, Alberta Environmental Protection, personal communication). An increasing number of private land owners are also considering managing their forested land for its timber value rather than solely for its agricultural value.

The downside of private forests being used for their fiber value for economic gain is the potential for poor management with significant negative impacts on water quality, biodiversity or other concerns. This is a real risk since there are few, if any, regulations governing the harvest of private forests in most Alberta municipalities (B. Grundberg, 1996, Alberta Environmental Protection, personal communication). In order to offset this risk, it may be wise for senior governments to create a model which would outline different kinds of sensitivity of forested areas and identify levels of risk associated with forest harvesting. For example, GIS could be used as a tool to create maps which would overlay forest parcel locations, soil erodibility, and proximity to streams, in order to identify areas for special attention if logging is to take place.

Considerable interest in the woodlot industry has developed over the last 2–3 years in Alberta as demonstrated by the emergence of a provincial woodlot association and significant attendance at a number of woodlot information meetings (J. De Franceschi, 1996, Canadian Forest Service, personal communication). There is also a demonstrated need for more information to be in the hands of private forest holders and resource agencies. A completed inventory would provide the basis for resource agencies and local governments to create effective guidelines, programs, and technical assistance needed to make a private woodlot industry sustainable.

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Interrelationships of Streambed Gravel, Bedload Transport, Beaver Activity and Spawning Sockeye Salmon in Stuart-Takla Tributaries, British Columbia, and Possible Impacts from Forest Harvesting

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Abstract

The Stuart-Takla Fishery/Forestry Interaction Project was initiated to provide a comprehensive understanding of the watershed processes that could be used to effectively manage the natural resources of north-central British Columbia. Objectives, techniques and methods are discussed, and results from 1990-94 are presented here for the salmon spawning gravel and bedload movement studies of this project. Freeze-core samples for Gluskie, Forfar, Kynoch, and Bivouac creeks indicated that a typical armoring layer of coarser gravel was only present on the surface of the streambeds during June and July, because sockeye salmon (*Oncorhynchus nerka*) spawning during July and August scoured fine particles from the bed and mixed the gravels that had been sorted during the May–June floods. Streambed sediments were coarser in Gluskie Creek and in upstream sections of the other creeks where gradients were steeper. This spatial variability of sediment size produced an inverse correlation between geometric mean particle size of the bed and depth of salmon redds ($r = -0.50$). The positive relationship each year between mean particle size of the streambed over a redd and adjacent to it indicated that spawning salmon substantially altered gravels by removing fine particles (mean intercept = +8.3) and that they conditioned areas of small gravel at nearly twice the rate of areas with coarser gravel (slope = +0.71). Most transport of streambed sediment occurred as bedload during May and June (snowmelt) and August (salmon spawning), with salmon producing 10-39% of this transport. The number of spawners was also related to the volume of bedload transport during August ($r^2 = 0.50$). Beaver dams across Kynoch Creek caused fine sediments to settle on the surface of the streambed; these fine sediments reduced the incubation success of salmon eggs and alevins during 1992, 1993 and 1995. A complex interrelationship has evolved among the physical and biological processes of these ecosystems, which must be understood and quantified before the effects of forest harvesting can be understood.

Scrivener, J.C., and Macdonald, J.S. 1998. Interrelationships of streambed gravel, bedload transport, beaver activity, and spawning sockeye salmon in Stuart-Takla tributaries, British Columbia, and possible impacts from forest harvesting. Pages 267-282 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1-4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

British Columbia (B.C.) watersheds produce valuable renewable resources such as timber, salmonids, wildlife, and grasslands. Resource production within these ecosystems is so interrelated that harvesting one often affects the others (Hartman and Scrivener 1990; Meehan 1991; Naiman 1992). Effective and successful management of these public properties requires a comprehensive understanding of the physical and biological processes in the ecosystems. In B.C., most previous research has been conducted in productive coastal forests including the Carnation Creek Experimental Watershed Project on Vancouver Island (Poulin and Scrivener 1988; Hartman and Scrivener 1990), the Fish Forestry Interaction Program (FFIP) in the Queen Charlotte Islands (Chatwin et al. 1991) and a synoptic survey in Barkley Sound (Brown et al. 1987). Comprehensive Coastal Fish/Forestry Guidelines were developed from this knowledge (British Columbia Ministry of Forests and Lands et al. 1992). Intensive studies of forest herbicide use have also been concluded in coastal areas (Reynolds et al. 1993). Research in the interior of B.C. has been limited to areas such as Slim Creek (Slaney et al. 1977; Choromanski et al. 1993). Coastal trees, soils, geology, climate and salmonid habitats are very different than those of sub-boreal forests of the central interior (Bustard 1986; Hartman and Scrivener 1990; Macdonald et al. 1992).

The Stuart-Takla Fishery/Forestry Interaction Project (STFFIP) was initiated during 1990 to provide research knowledge of ecosystem processes so that the Forest Practices Code and Fishery/Forestry Guidelines could be improved for the interior of B.C. (British Columbia Ministry of Forests and Lands et al. 1992). This project is an integrated, multi-disciplinary, long-term, and multi-watershed study, whose results promote an understanding of ecosystem complexity as it relates to salmonid, timber and wildlife production (Macdonald 1990). The streambeds of Bivouac, Gluskie, and Kynock watersheds (36–76 km²) are being studied during pre-logging (4+ yrs) and post-logging periods, while Forfar watershed (36 km²) is being studied in its pristine state. A large population of sockeye salmon (*Oncorhynchus nerka*) uses the streambeds of these watersheds for spawning from late July to early August and for incubation of their eggs from August to May. During August or September of some years, beaver (*Castor canadensis*) established winter territories by building low dams across the flood plains of these small streams. Pools developed that extended 200 to 300 m upstream, covering some of the salmon redds, flooding the surrounding area of deciduous riparian vegetation, and providing beaver runs and food storage under winter ice and snow. These beaver dams are usually washed away when melting snow produces flood conditions during the following spring.

Most gravel streambeds consist of a matrix of finer material that occupies part of the voids between a framework of coarse particles (Church et al. 1987). Salmonid eggs and alevins can incubate in these voids when adequate space remains for water exchange and alevin movement (Dill 1969). Typically, framework particles are in contact with one another and form a stable, self-supporting structure, while matrix particles form 20–30% of the remaining total weight (Church et al. 1987). Alluvial gravels become matrix supported when the matrix exceeds 30%, because all voids are filled and framework particles become separated from one another by matrix. Matrix material is missing from the surface of most streambeds. The coarse surface layer of interlocking framework material is usually 1–2 particle diameters thick and it is formed either by the scour of matrix particles from the surface framework (Sutherland 1987), or by the relatively immobile large particles becoming concentrated on the surface during deposition (Andrews and Parker 1987).

The natural physical and biological influences on streambed gravels in the study streams must be understood before any impacts from forest harvesting can be assessed. Samples of the streambed and transported bedload from 1990 to 1994 are used here to characterize pre-logging gravels of these streams. The dynamics of the streambed are then shown in relation to the hydrological regime, spawning sockeye salmon and beaver activity. Comparisons with data from other streams indicated some unusual features in the STFFIP tributaries. The adequacy of our sampling methods to detect possible changes from forest harvesting is also discussed.

Methods and Materials

Since 1990, samples of the streambed have been obtained from tributaries of Takla Lake and Middle River (Fig. 1). Ten samples were obtained and maps drawn at each of four sites in the lower 1500 m of Gluskie, Forfar, and Kynock creeks during late September each year. Samples were also obtained from Forfar and Kynock creeks during July 1994, after the spring flood but before salmon spawning. Signs were established at 50-m intervals along each

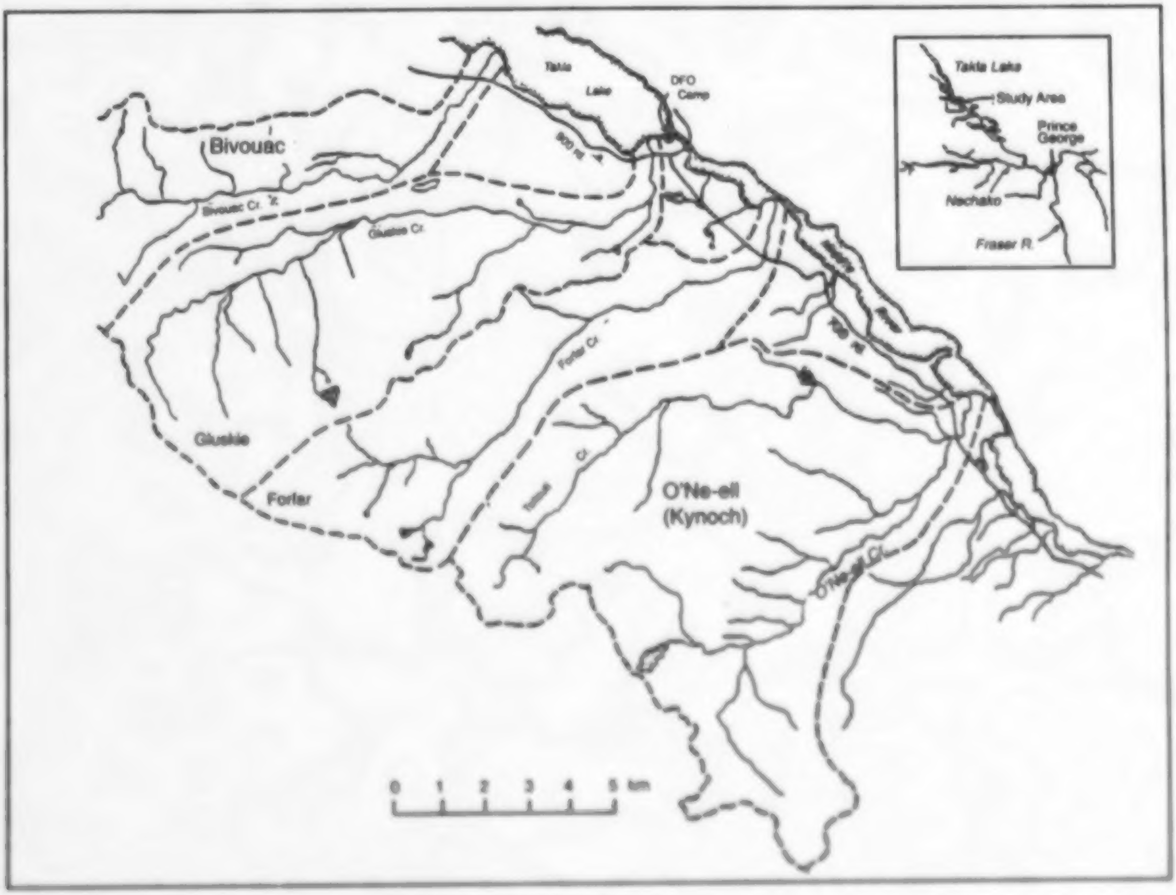

Figure 1. Map showing four experimental watersheds at Takla Lake, some of the stream study sites, the fisheries camp (DFO), and logging access roads.

stream using the lowest site for the adult salmon fence as 0 m and hip-chain measurements along the thalweg of each stream. Permanent survey hubs (20-30) were established at <5-m intervals at three about 80-m reaches in each stream. They were used to determine annual changes of channel morphology and large woody debris (Hogan 1990). Other survey hubs were also established to trace the movement of magnetized bed material during freshets and salmon spawning (Gottesfeld 1990). Many streambed samples were obtained from these channel cross sections. The signs, annual bedload maps and the survey hubs were used to locate the same sample sites every year.

Additional streambed samples were collected during 1992, 1993, and 1994 when beaver built dams at 100, 510, and 580 m, respectively, on Kynoch Creek. Samples were obtained from the shallow pools upstream of the dams (10-70 m) and from sites below the dams. They were compared with samples from 300, 650, and 700 m on Kynoch Creek and from 150, 260, and 300 m on Forfar Creek.

A modification of Ryan's (1970) freeze-core technique was used to sample the streambed. A 5-cm diameter steel probe attached to a sampling pot was driven 30 cm into the bed (Scrivener and Macdonald in press). Acetone and dry ice were added to the pot. The acetone, cooled to -60°C and circulated in the sampler, froze a 25-30 cm diameter gravel core around the probe in 25 to 40 minutes (depending on stream temperature). The frozen cores, weighing 15 to 25 kg, were split into top (0-15 cm) and bottom (15-30 cm) layers; any rocks that protruded >50% outside the core diameter and that were not against

the probe were discarded. They were representative samples of the streambed because core diameters were usually >2-fold larger than diameters of the largest particles (mean = 78 mm for 190 cores) and because excluded rocks prevented excessive representation of these particles in the samples (Lotspeich and Everest 1981; Shirazi et al. 1981). Each layer was placed in a labelled polyethylene ore bag and each core placed in a burlap sack for transport to the laboratory (Scrivener and Brownlee 1989). A few samples were discarded when the probe became frozen to a buried log. Most of the sediment on these cores had been lost before the sampler could be extracted from the streambed and therefore new cores were obtained.

When a core contained salmon eggs, the number of eggs was categorized as a few (1–8), scattered (about 12), or a distinct redd (>20). The depth range of egg clusters was measured to the nearest 0.5 cm with a metre stick as the freeze-core was split and bagged. During early October, a shovel and collecting net were also used to sample salmon eggs from redds within beaver ponds and from redds above and below the ponds. In the field, the proportion of live (clear-eyed eggs) to dead eggs (opaque) was calculated for each sample and most samples were preserved in Stocker's solution.

Data from coastal streams were obtained and treated in a similar manner (Scrivener and Brownlee 1989; Hartman and Scrivener 1990). We had obtained freeze-cores annually from Carnation Creek between 1973 and 1989 and they were compared with the STFFIP samples.

In the laboratory, each layer was oven dried at 105°C. The mean diameter and weight of the largest rock were obtained before the layer was passed through a sample splitter with 24 mm dividers. Particles too large for the splitter were weighed; after 1990, the weight of these particles retained on a 50-mm sieve was also determined. Split portions of the sample were weighed and one portion was passed through five nested sieves (9.35, 2.38, 1.19, 0.298, and 0.074 mm) using a Ro-Tap Model T-674 testing shaker (Scrivener and Brownlee 1982). Separated components were weighed and all weights were entered onto the VAX computer database at the Pacific Biological Station, Nanaimo, B.C. This analysis separated particles into medium and small cobbles; large, medium, and pea gravels; coarse, medium, and fine sands; and silt-clay components as classified by the U.S. Department of Agriculture (Scrivener and Brownlee 1982).

Geometric mean particle size (*Dg*; Platts et al. 1979) was calculated for each layer of each freeze-core using

Equation 1 $Dg = d_1^{w1} \cdot d_2^{w2} \cdot \cdot \cdot \cdot \cdot d_n^{wn}$

Where d = the geometric mean diameter between two adjacent sieve sizes and w = the proportion of the layer retained by the smaller sieve. The largest "d" was calculated using the diameter of the largest rock and the size of the nearest smaller sieve. When this rock was >100 mm, the equation was expanded using the rock's weight and size to calculate an additional "d" and "w". When it was <50 mm, an element was dropped from the equation.

A cumulative particle size distribution of each layer was calculated as the sample percentage smaller than each size category. Each distribution was fitted to

Equation 2 $Percent = a + b^{*}\log_{10}SIZE$

using Shirazi's technique (Shirazi et al. 1981). Where Percent = the inverse probability transformation that the percentage of the sample is smaller than a given mesh *SIZE*, *a* = intercept of the regression line, *b* = slope of the line, and *SIZE* = mesh size in mm. These equations were statistically significant representations of the distributions even with only nine or ten points (If n = 9, P <0.05 when R2 >0.45). They also predicted >80% of the variability for 695 of 960 distributions (mode $R^2 = 0.83$). Twelve equations had $R^2 < 0.70$ and most of these samples contained exceptionally large rocks (>128mm). They were recalculated and the R^2 value improved after excluding the weight of the large rock. Thus, these equations could accurately predict the percent of sample finer than any mesh size.

A Fredle Index (*Fi*; Lotspeich and Everest 1981) was calculated for each layer of every freeze-core using

Equation 3 $$Fi = \frac{Dg}{\sqrt{d_{75}/d_{25}}}$$

where *Dg* was calculated using Equation 1, and d_{75}, d_{25} were calculated as particle size diameter at the 75% and 25% percentile, respectively, using Equation 2. Thus, *Fi* provided a single measure incorporating mean particle size in a layer and associated variation of particle sizes around the mean.

Transported bedload was collected in four to six empty buckets that were buried in each of Gluskie, Forfar, Kynock, and Bivouac creeks. The 20-L plastic buckets were buried along a single cross section near the stream's mouth (station 50-100 m; Fig. 1). The

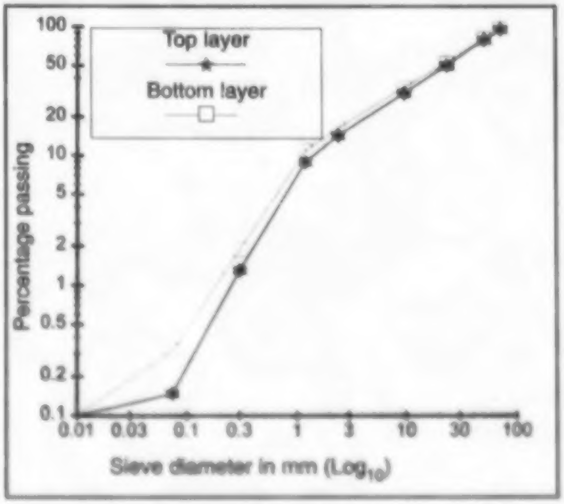

Figure 2. Mean distribution of particle sizes in the top (0–15 cm) and bottom layers (15–30 cm) of 178 freeze-core samples that were collected from Forlar Creek during 1990 to1993. Both axes have a Log_{10} scale.

plastic lids of the buckets were level with the streambed and they contained 5 × 10 cm cut slots until June 27, 1993 and 7 × 7 cm cut slots thereafter. Every few days during periods of rapid bedload movement, depth measurements of the sediment that settled in the buckets were obtained. Notes were made of any sediment deposited on top of the lids or any erosion from around the buckets. After each erosional event, a bucket's elevation was adjusted or the bed around it was levelled if the lid was no longer flush with the streambed. Each bucket was lined with a polyethylene ore bag so that collected sediments could be easily removed. Any contents were emptied five times each year and a sub-sample was saved for particle-size analysis using the same procedures described for freeze-core samples. Sampling periods coincided with the end of winter (April 20–26), the end of spring freshet (around June 30), the beginning of sockeye spawning (July 25–31), the end of sockeye spawning (around August 15), and the beginning of winter (October 2–10). Volumes collected in the buckets were extrapolated to the total channel width, and total volumes transported were calculated for each channel cross section.

These studies of bedload movement were designed to complement the magnetic rock studies by [illegible] (1998). We quantified the particle size distribution of bedload and the total volume passing a specific cross section of each stream. Movements of magnetic rocks are used to quantify distance moved and the deposition pattern of bedload in relation to the size of individual particles.

Results and Discussion

Streambed Characteristics

The streambeds of our study streams contained framework-supported gravels. Particles of framework size, >24 mm in diameter, formed >50% of freeze-core weights, while matrix particles (<2.38 mm) were <18% (Fig. 2). That is, framework particles were likely in overlapping contact, since matrix was <30% of most freeze-cores, and matrix particles likely did not fill the voids within the framework, since matrix is <20% of a freeze-core (Church et al. 1987), and thus space must be available for incubating salmonid eggs and alevins, and for water circulation. A similar pattern was observed for our coastal comparison (Hartman and Scrivener 1990).

Both large and small particles were rare in STFFIP tributaries. Medium cobbles (dia. >100 mm) were collected only occasionally in Forfar (Fig. 2) and Kynoch creeks. They were slightly more common in Gluskie Creek (Scrivener and Macdonald 1996). Particles of fine sand, silt, and clay (<0.3 mm) usually formed 1–1.8% of the freeze-cores (Fig. 2). Silt and clay particles (<0.074 mm) comprised <0.3% of the samples and they were very rare in the top 15 cm of the streambed. The rarity of these fines was surprising as there are large areas of lacustrine deposits within each watershed (Ryder 1995). Comparable areas from 0.2 to 4.0 km in Carnation Creek had similar quantities of fine sand, but more silt and clay (Fig. 3). This was also surprising because coastal soils are shallow with fewer sources of these fines and because coastal soils and streambeds are frequently scoured during the numerous winter freshets (Hartman and Scrivener 1990).

Typically, riverine gravels have a coarse surface layer of 1 to 2 particle diameters thickness (Church et al. 1987), and a distinct layered pattern (Carling and Reader 1982; Scrivener and Brownlee 1989), but these characteristics were rare in STFFIP tributaries. This surface armoring layer consisted of structurally interlocking particles that resisted stream flow and tended to stabilize the bed. Large keystone particles must be moved by scouring flows before the bed can be mobilized (Reid and Frostick 1984). This surface layer typically produced [illegible] values in the top layer that were [illegible] of those in the bottom layer of freeze-cores (Fig. [illegible]). In contrast, [illegible] freeze-cores from September samples showed few size differences between top

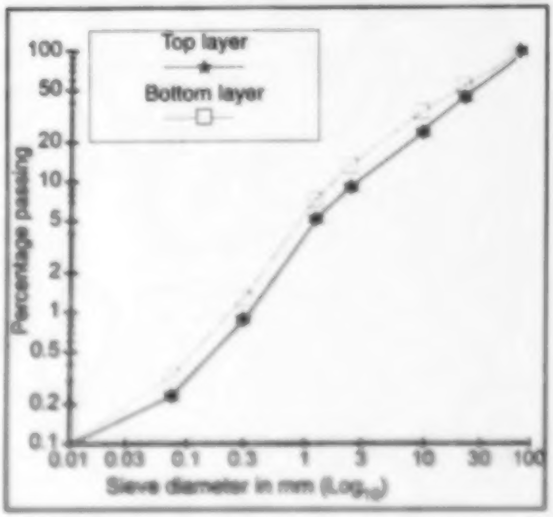

Figure 3. Mean distribution of particle sizes fromthe top (0–15 cm) and bottom layer (15–30 cm) of 384 freeze-cores from 0.2 to 4.0 km of Carnation Creek (a coastal stream) collected between 1973 and 1981, before any logging impacts were observed. A $\log_{10}$ scale is used for both axes.

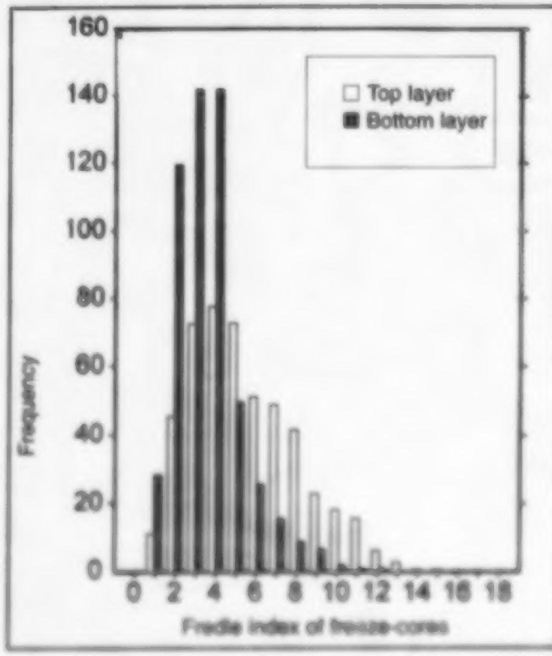

Figure 4. Distribution of Fredle indices from the top and bottom layers of freeze-cores collected from an area of intense spawning by chum salmon in the upper estuary of a coastal stream, Carnation Creek, 1981 to 1985.

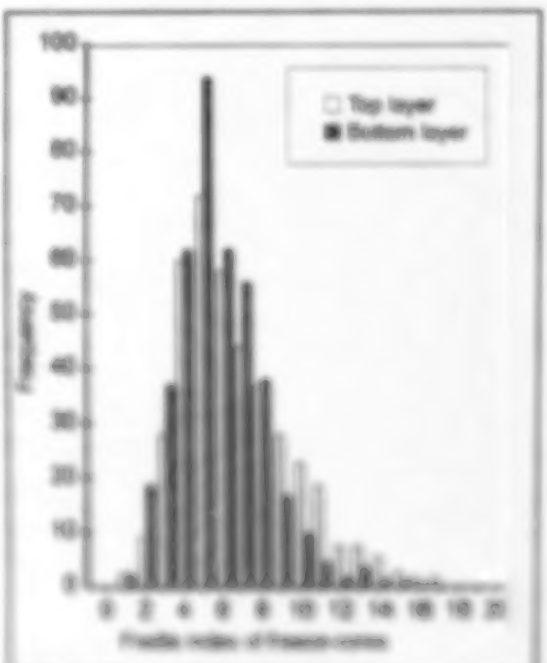

Figure 5. Distribution of Fredle indices obtained the top and bottom layers of freeze-cores cores collected from Forfar and Kynoch Creek each September from 1989 to 19[illegible].

and bottom layers (Fig. 5). The armoring layer in Carnation Creek was evident for both the particle size (Fig. 3) and Fi distributions (Fig. 4). Fine gravel (<9.55 mm), coarse sand (<2.38 mm), and medium sand (<1.19 mm) also did not increase with depth in the STFFIP tributaries (Fig. 2) as they did in Carnation Creek (Fig. 3). STFFIP freeze-cores from July samples had surface layers that were slightly coarser than bottom layers (Fig. 6) indicating that at least a thin armoring layer existed after the streambed was influenced by snowmelt peak flows. A thin armor layer was developed by the hydraulic processes of the June floods, but it was destroyed during August by the digging activity of spawning sockeye salmon (Fig. 5). Presumably, the streambed remained well mixed throughout most of the year (August through May) when freshets were rare.

Spatial differences in the streambed were also observed among the STFFIP tributaries. Freeze-cores from Forfar and Kynoch creeks appeared similar, but Fi values were smaller than those of Gluskie Creek (Fig. 7). The distribution from Gluskie Creek was shifted towards the larger values probably because medium cobbles occurred more frequently

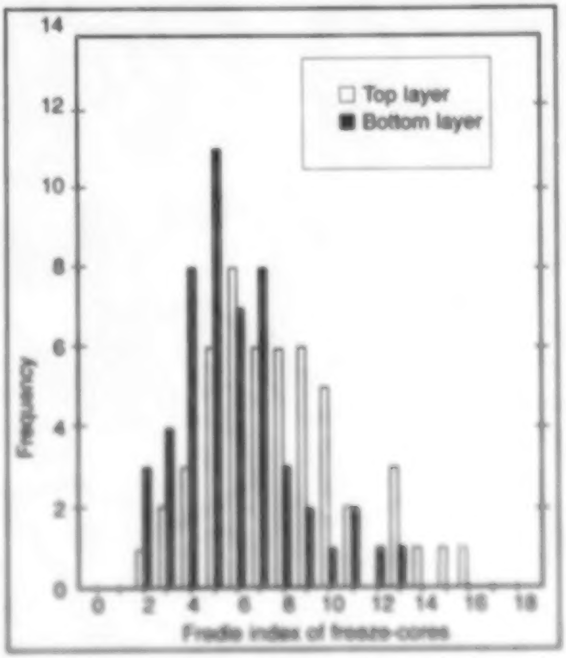

Figure 6. Distribution of Fredle Indices obtained for top and bottom layers of freeze-cores that were collected from Kynoch and Forfar creeks during July 1994.

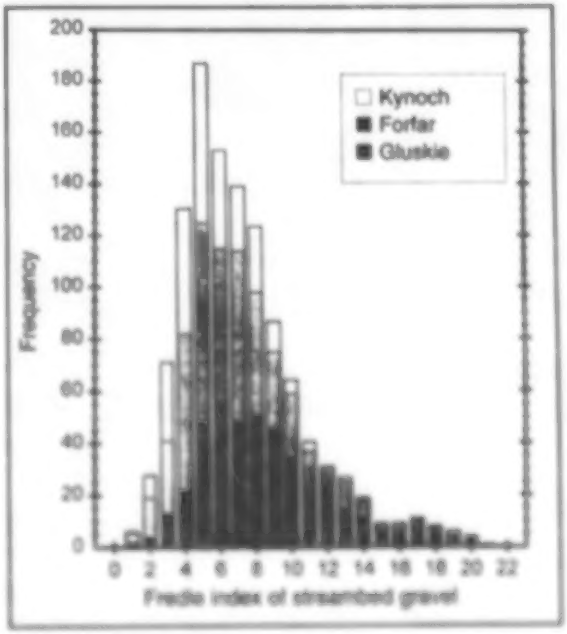

Figure 7. Fredle Indices from both top and bottom layers of freeze-cores obtained from Kynoch, Forfar and Gluskie creeks, 1990 to

in their freeze-cores (Scrivener and Macdonald 1998), but *Fi* values were still greater for Gluskie Creek when these extreme values were excluded (Wilcoxon two-sample test, P<0.001). The streambed also became coarser with distance upstream in the channel from the mouths to the road bridges. The *Fi* mode increased from 6.0 at the delta to 6.5 at 1500 m on Kynoch Creek and from 6.5 at 300 m to 7.0 at 1370 m on Forfar Creek (Wilcoxon two-sample test, P<0.001). Gradients in Forfar and Kynoch creeks increased approximately 0.5% between the mouth and the sampling sites 1 to 1.5 km upstream. The gradient increased more quickly in Gluskie Creek because topography confined the channel within 0.6 km of the mouth. Channel gradient is often directly correlated with median diameter of the bed material and steeper reaches often have a capacity to transport more sediment than is supplied to them (Simons and Simons 1987). Sediments are also stored at depths in fluvial fans near the mouths of streams (Dey 1987; Reid and Frostick 1994). Thus, upstream channels are zones of degradation and erosion that leave coarse substrates, as fine material is transported downstream where large clasts can be buried in deep sediment deposits (Church et al. 1987).

Streambeds in the STFFIP watersheds provided better incubation substrates for salmonid eggs than most of those reported in the literature. The median graphic mean diameter (D_g) was 22 mm for 120 different spawning gravels (reviewed by Kondolf and Wolman 1993), while it was 30 mm for this study. Half of the literature means were between 10 and 27 mm for graphic means, and between 8.7 and 28 mm for geometric means. The geometric mean was <22 mm for Takla spawning sediments and particles >100 mm in diameter were rare. Particles >100 mm were commonly found in 63 of the 120 spawning gravels used by spawning Chinook salmon (Kondolf and Wolman 1993). These characteristics could explain why the Takla tributaries produced survivals from potential egg deposition to fry emergence of 16-20% (G. Smith, DFO, New Westminster, B.C., personal communication), while survivals of 5-15% are reported for other interior and coastal populations of sockeye salmon (Foerster 1968).

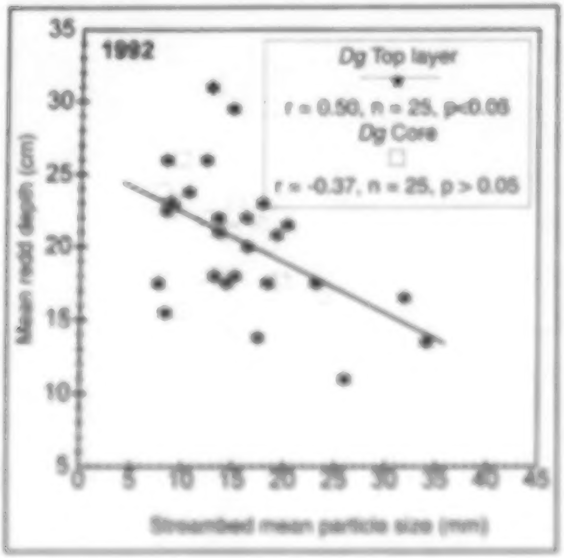

Figure 8. Relationships observed between mean depth of a salmon redd and geometric mean particle diameter (*Dg*) of the top layer and total freeze-core containing sockeye salmon eggs from Kynock, Forfar and Gluskie creeks during 1992.

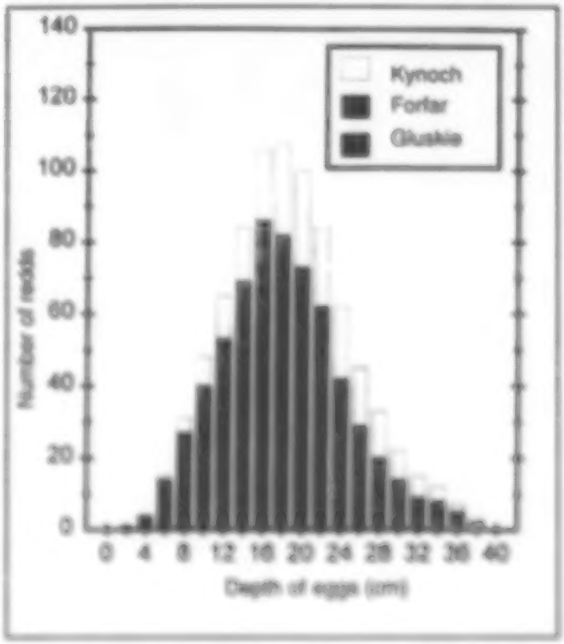

Figure 9. Distributions of egg depth for 191 sockeye salmon redds from Kynock, Forfar and Gluskie creeks. Individual redds were distributed over 4-10 cm in 1990-94 freeze-core samples.

Spawning Sockeye Salmon

Streambed characteristics influenced the depth of sockeye salmon redds in the STFFIP tributaries. Mean depth of a redd was negatively correlated with *Dg* of the top layer of the freeze-core, but a significant correlation was usually not obtained for the total freeze-core (Fig. 8). This relationship persisted for every year of the study, but variability was greater during years with large numbers of spawning sockeye salmon (r= [illegible] to [illegible]) than during years with fewer spawners (r= [illegible] to [illegible]). Redds with a *Dg* <10 were rare during years with large escapements, while redds with a *Dg* <20 were very rare during years with a small escapement (Scrivener and Macdonald 1998). These results possibly indicate that spawning sockeye salmon clean fine particles from the streambed covering the redd and that they avoid coarse spawning gravels when low numbers permitted site selection. Because spatial differences existed for streambed particle sizes, the spatial pattern of redd depths was similar to particle size. Redd depths were <3 cm shallower in Gluskie Creek than the other streams (Fig. 9) and they tended to become shallower with increasing distance upstream from the mouth to the road bridges (Scrivener and Macdonald 1998). This probably indicates the coarsening of the top 30 cm of the streambed as stream gradient increased.

Among spawning salmonids, most variability of redd depth has been related to female size. Species with larger individuals tend to use coarser spawning gravel and they bury their eggs more deeply in the streambed (Van den Berghe and Gross 1984; Crisp and Carling 1989; Kondolf and Wolman 1993). Within a species, female size differences explained [illegible] of the variability of redd depth among trout and Atlantic salmon (*Salmo salar*) in streams of England and Wales (Crisp and Carling 1989), and among coho salmon (*O. kisutch*) in a Canadian stream (Van den Berghe and Gross 1984). In these studies, [illegible] of redd depth variability was explained by differences of streambed composition. However in STFFIP tributaries, [illegible] of the spawners are of a single age group (H. Smith 1994) and of similar size (SD = [illegible] for 4-yr-old females from the same stream) and mean length differences < [illegible] cm between streams during any year (L. Smith 1997). Now

Westminster, B.C., personal communication). Gravel size has an impact on the ability of a female to construct a redd. The significance of this relationship declined during years with high spawner density probably because they were forced to use sites that were normally avoided and because they interfered with each other's activity.

In an attempt to quantify spawner influences on streambed characteristics, freeze-cores containing redds (>20 eggs) were paired with those from nearby sites without eggs. Undisturbed sites adjacent to redd sites were selected from a location immediately upstream (1–4 m), immediately to one side (about 1 m), or were calculated as the mean of the two. Some redds were clustered so freeze-cores from a few undisturbed sites were paired with more than one redd, and a few of the redds were dropped from the analysis because an adjacent undisturbed site that fit the criteria did not exist.

Linear regressions between the *Dg* of redds and of adjacent undisturbed sites explained 35–45% of the variability in the streambed during all years except 1993 (Fig. 10 a–d). Here, the regression was still statistically significant because so many redds were found, but the R2 value was of little value as a predictor of change (Fig. 10d). Slopes of the annual regression lines were steep (near 1.0) during 1990 and 1992, years of small escapements, indicating a consistent removal of fines from all redd types (Fig. 10 a,c). Slopes were <0.63 during years containing large escapements indicating that more fines were removed from redds with small particles than from redds with large particles (Fig. 10 b,d).

Modification of fluvial gravel size by spawning salmonids has been discussed extensively, but it has rarely been observed in the field except immediately after spawning. Previous studies have shown both streambed modification after spawning and no difference between redds and adjacent undisturbed sites (reviewed by Everest et al. 1987 and Kondolf et al. 1993). Among studies that showed streambed changes, particles <1 mm in diameter (clay, silt, and fine and medium sands) were reduced [illegible]%, while particles <2 mm (all sands) were reduced only 15–30% immediately after spawning. In the STFFIP tributaries, reductions ranged from 20–50% for particles <1.19 mm and from 10–40% for particles <2.38 mm. Other studies have shown little or no change in mean particle size of the streambed (Everest et al. 1987; Kondolf et al. 1993).

These conflicting results might indicate that fine sediments rapidly reinfiltrated salmonid redds. Surface sediments are disturbed by freshets and hydraulically winnowed so that they are coarser than sediments at depth (Carling 1984). This occurs during and immediately after autumn spawning in coastal streams (Crisp and Carling 1989; Scrivener and Brownlee 1989), so any salmonid effect would be quickly masked. Reinfiltration rates of fine sediment indicated that trout redds could be fully silted within a few days in streams with typical loads of suspended sediment (Carling and McCahon 1987); thus, influences of spawning usually rapidly disappear from coastal streams. In STFFIP tributaries, redds are subjected to clear base flows, they are covered by ice and snow, and there are few predators. These modified spawning gravels would persist unchanged for months, permitting sockeye salmon to thrive in watersheds that are rich in fine sediments (Cheong et al. 1995; Sanborn 1996). Long-term effects of spawning were recorded for salmonid streams without winter freshets in Arizona, California, Idaho, and central BC (Kondolf et al. 1993).

Bedload Characteristics

Sediment is transported in streams as a function of discharge and rate and size of the sediment supply. It moves either within the water column as suspended load or by bouncing or rolling (saltating) along the streambed as bedload (Hey 1987). Rates of suspended transport are usually calibrated to stream flow or discharge (Cheong et al. 1995), but rates of bedload transport are more episodic and vary markedly with time even at similar flows (Hubbell 1987).

During both 1991/92 and 1992/93 water-years, 80–89% of bedload transport occurred when the Takla tributaries were in May/June freshet and when sockeye salmon were spawning (Fig. 12; Scrivener and Macdonald In press). Transport rates and total volumes were often proportional to watershed area during the period of snow-melt (Scrivener and Macdonald In press), but they were not proportional to area during spawning (Fig. 11). This pattern persisted even though 1992/93 produced nearly twice the transport volumes during a much shorter time period than were observed during 1991/92. Spawning salmon caused 20–50% of the movement of bed material in Kynock, Forfar, and Gluske creeks (Fig. 11). In Bivouac Creek, little movement occurred during August 1992 when only 81 spawners entered the creek, but this changed the following year when 20 000 spawners were observed (Fig. 12).

The number of spawning sockeye salmon was regressed against the volume of bedload transported in the four study streams. With combined data from

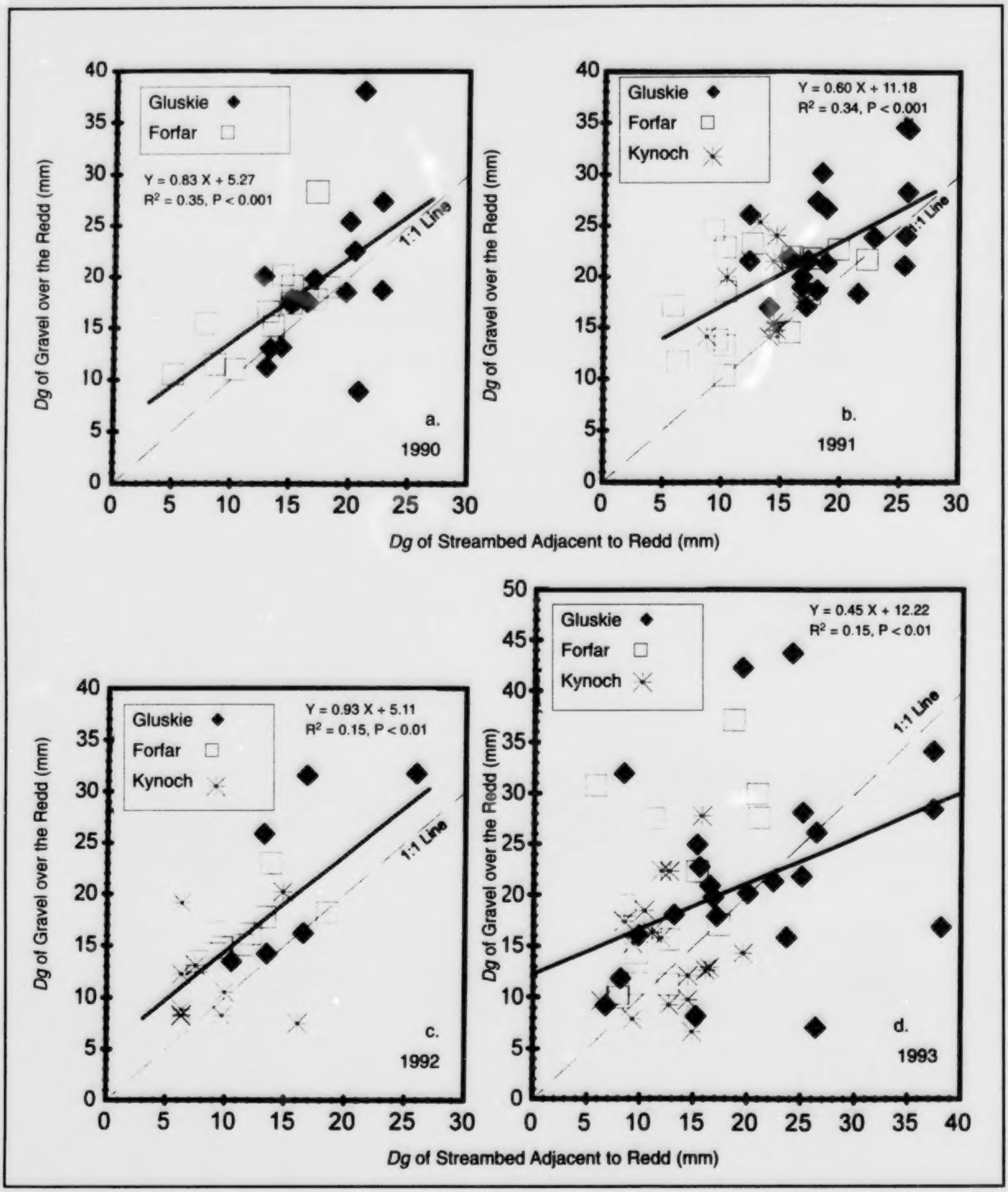

Figure 10. Relation between geometric mean particle diameter (*Dg*) for spawned and undisturbed gravels in Gluskie, Forfar, and Kynoch creeks during a) 1990, b) 1991, c) 1992, and d) 1993. Regressions are shown as solid lines, while dashed lines would represent 1:1 ratios.

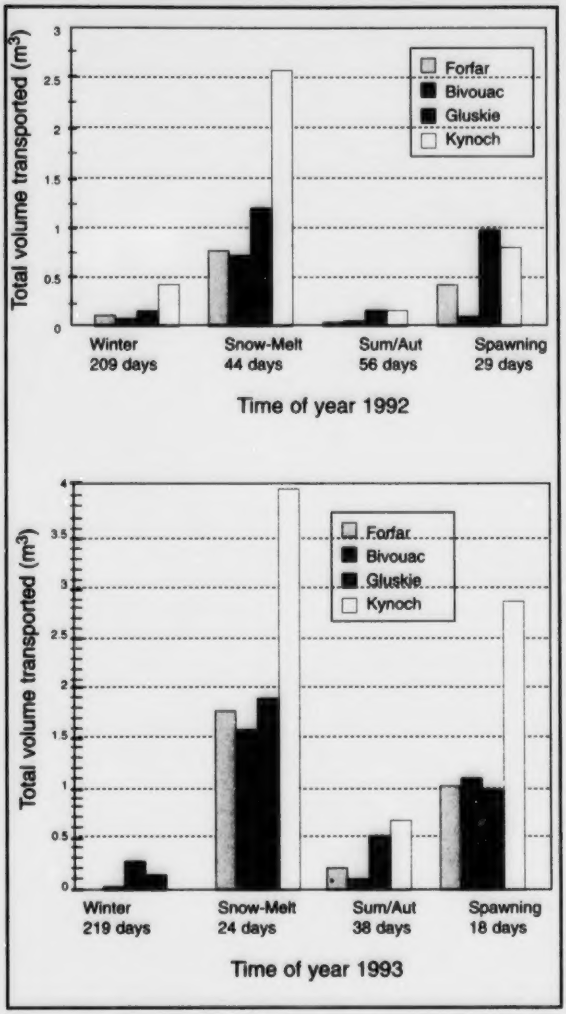

Figure 11. Total bedload volumes transported in Forfar (36 km²), Bivouac (34 km²), Gluskie (47 km²), and Kynoch (76 km²) Creek watersheds during the periods of winter (October to May), spring snowmelt (May and June), summer/autumn (July and September), and sockeye spawning (August). The data is arranged by hydrological water-year, which begins October 1 and ends the following September 30.

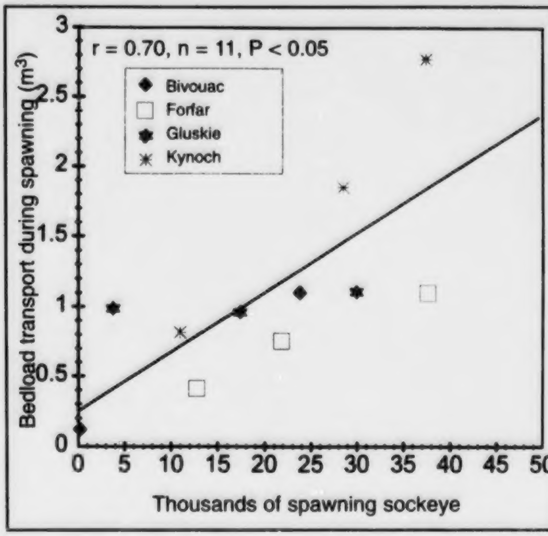

Figure 12. Relationship observed between total volume of bedload transport during the spawning period and number of sockeye salmon spawning in Bivouac, Forfar, Gluskie, and Kynoch creeks during 1991, 1992, and 1993 (data from Smith 1994).

the four streams, the regression was significant, but it was of little predictive value because it explained only 50% of the variability in bedload transport (Fig. 12). The volumes moved were greater in the largest creek, Kynoch, than in the smallest creek, Forfar, but the same trend was apparent for both streams. This trend was not as apparent for the data collected from Gluskie Creek. Here, the 0.3-km long delta area contained the best spawning habitat which appeared to be saturated even during years of relatively low escapement (Tschaplinski 1994). Upstream areas in Gluskie Creek contained larger streambed materials (Fig. 7) that probably made redd excavation more difficult. When more data are available in future years, a separate analysis could be obtained for each stream. A similar relationship has not been reported elsewhere. Most previous studies were done in coastal watersheds where spawning occurred during flood conditions (Hartman and Scrivener 1990; O'Leary and Beschta 1981) or where spawner densities were low (Van den Berghe and Gross 1984; Crisp and Carling 1989).

The distribution of particle sizes that were transported as bedload varied with the season. Our bedload buckets contained only 22% sand, silt and clay-sized particles (<2.38 mm) when salmon were spawning during August, but these particles formed >65% of the bedload during the winter (Fig. 13). Distributions of bedload particle sizes were also similar to those of the bottom layer of freeze-cores collected during the spawning period (Fig. 2 and 13) indicating that all particle sizes were being moved and that they were moved in proportion to their

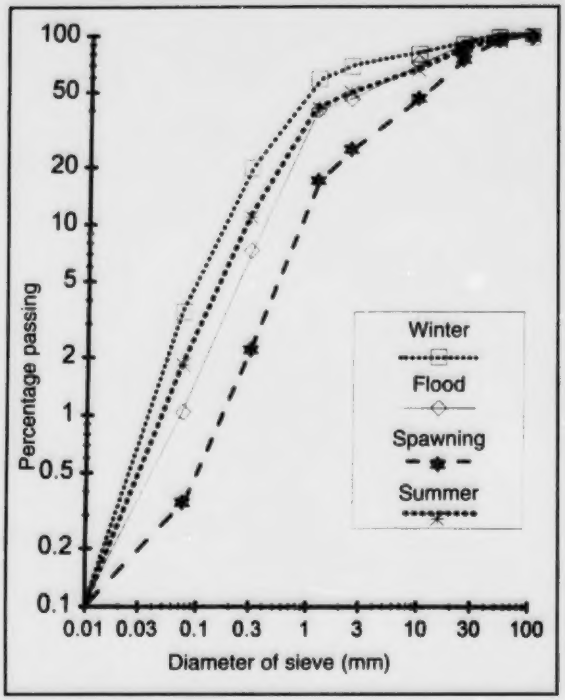

Figure 13. Mean distribution of particle sizes in the bedload buckets during the winter (October to May), spring flood (May and June), salmon spawning (August), and summer/autumn periods (July and September) for 1991, 1992 and 1993.

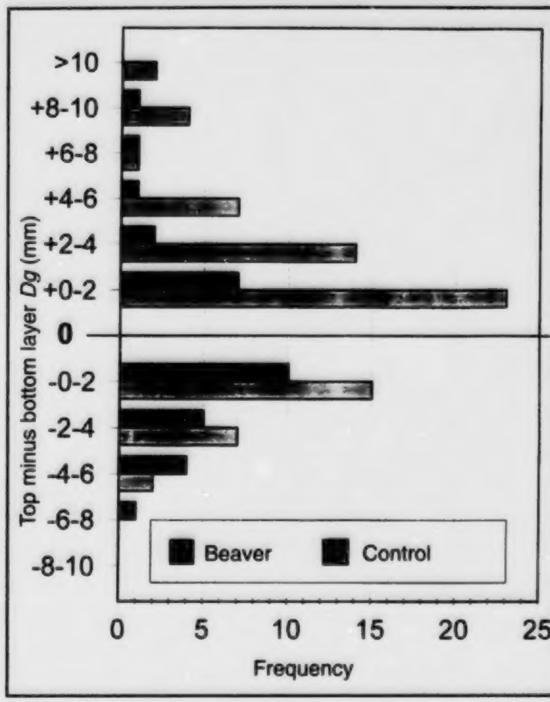

Figure 14. Difference between geometric mean part icle size of top and bottom layers (Dg_T - Dg_B) of frozen cores from beaver ponds in Kynoch Creek and from control reaches in Kynoch and Forfar creeks. Freeze-core samples were collected during September of 1992, 1993, and 1995.

presence in the streambed. Distributions from the spring snowmelt and summer/autumn periods were intermediate, with the majority of particles being larger in size than coarse sand (>2.38 mm; Fig. 13). Our summer/autumn samples might be atypical and contain coarser material than usual because the buckets were contaminated by a few late spawners during August 1991, and because two unusual freshets moved substantial bedload during July 1993.

The water-years, 1991/92 and 1992/93, were good indicators of the lower and upper range of bedload transport and spawning activity in these tributaries. A small snow pack and dry spring during 1992 produced a moderate spring flood and moderate bedload movement (Macdonald et al. 1992). Poor marine survivals, substantial fishing, and warm migration temperatures in the Fraser River produced minimum sockeye salmon escapements and spawning activity during 1992 (Williams et al. 1992; Smith 1994). During 1993, a cool spring with rain-on-snow events generated peak floods with extensive erosion in the channels. Two July freshets and decade escapement maximums were also observed during 1993. Thus, bedload volume was three fold greater in 1993 than during the previous summer in our study streams.

Impacts of Beaver Dams

Beavers are common in forested areas of north-central B.C. They modify stream channel geomorphology and hydrology through the retention of sediment, organic matter and water, thus creating wetlands and meadows (Naiman et al. 1984). Wetlands provide rearing habitat for salmonids, but may affect spawning habitats (Meehan 1991). Within a few years, streams have retained as much as 6 500 m^3 of fine sediment and the wetted area of channel has increased about 200-fold behind each dam (Naiman et al. 1986). Water velocity over and

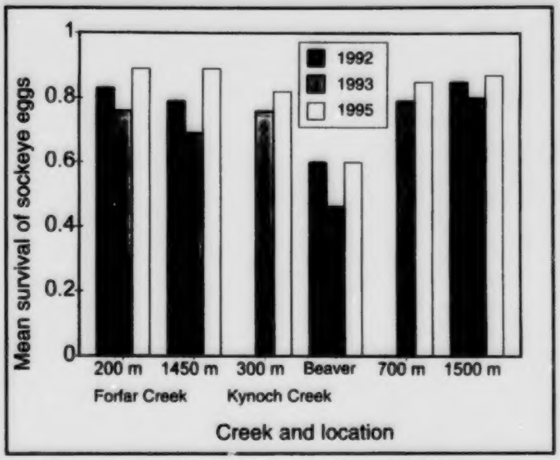

Figure 15. Mean proportion of live-to-dead eggs collected from various sites in Forfar and Kynoch creeks during late September of 1992, 1993, and 1995.

within the salmon redds is reduced and there is concern that the ponds may act as traps for fine sediment filling the pore spaces in the incubation gravels. Indeed, our 1992 data indicated that significant quantities of fine sand, silt, and clay had accumulated at the beaver affected site within four weeks of dam construction (Scrivener and Macdonald In press). The 1993 dam was built only 10 to 15 days prior to freeze-core sampling, so fine sediments were not significantly greater statistically at the beaver-impacted site than at control sites in Kynoch and Forfar creeks. Gluskie Creek data were excluded from the analysis because that creek's coarse gravels would bias our results.

The differences between *Dg* of top and bottom layers of streambed samples were compared for 1992, 1993 and 1995. Analyses of control-site freeze-cores showed that large particles were not concentrated on the surface of the streambed and that *Dg* was only slightly greater for the top layer than for the bottom layer (Fig. 5 and 14). In contrast, samples from the beaver affected sites had greater *Dg*s in the bottom layer than in the top layer (Fig. 14). Hence, fine sediments were accumulating on the surface of the streambed behind the beaver dams.

Two months after spawning, survivals of sockeye salmon eggs were also lower at the beaver affected sites than at any control site (Fig. 15). This pattern was consistent for all 3 years. Survival was also lower at all sites during 1993. The large escapement of sockeye salmon that year caused multiple spawning at many sites and continual interaction among spawning fish was observed. Salmon eggs are sensitive to disturbances during the first weeks of their incubation (Burgner 1991) and this probably increased mortalities at all sites.

Fine sediments began accumulating behind beaver dams in Kynoch Creek within a few weeks. Their impacts probably only remained for a single autumn and winter of egg incubation. Spring freshets and August spawners reconditioned the streambed by scouring fines from the incubation gravels. Suspended sediment levels in these streams were positively related to stream flow (Cheong et al. 1995) and concentrations were significantly greater during the spawning period (Scrivener and Andersen 1994a).

Beaver impacts on the fauna of aquatic insects have been observed, but little has been reported on survival of incubating salmonid eggs behind beaver dams. A 3-yr study in Algonquin Park, Ontario, indicated changes of community and emergence patterns, and reductions of density for aquatic insects when beavers constructed a dam across a small stream (Sprules 1940). During the next year, lotic species were replaced by species preferring slow-flowing silty habitats. Similar findings were obtained in foothill streams of the Rocky Mountains (Hodkinson 1975) and in Sept-Iles streams of Quebec (Naiman et al. 1984). Few direct measurements have been obtained for incubating salmonid eggs behind beaver dams (Meehan 1991). Most studies of beaver impacts on salmonids have been concerned with the expanded rearing habitat and with restricted adult access to spawning reaches upstream. Sockeye salmon in STFFIP tributaries spawned before the dams were constructed.

Ecosystem Complexity and Forest Harvesting

A complex interrelationship has evolved among the physical and biological processes of these ecosystems. They must be understood and quantified before forest harvesting impacts can be truly understood. Sockeye salmon populations in STFFIP tributaries have evolved mechanisms such as early spawning and development at very low temperatures that permit fry emergence during May before peak floods (Brannon 1987; Scrivener and Andersen 1994b); June rearing in swamps and flooded lands that are warm enough (>5°C) for early growth (Burgner 1991; Scrivener and Andersen 1994b); early

spawning that permits access prior to construction of beaver dams; mass spawning that conditions the incubation gravels (Fig. 10); and small alevins (28–30 mm) that easily move through streambed pore spaces (Dill 1969).

Evidence from soil maps, suspended sediment loads, bedload transport, and streambed composition indicated that sediment supply to our study streams would increase after forest harvesting is begun. Some of the deep till and lacustrine deposits that are common in the watersheds (Sanborn 1994; Ryder 1995) must be crossed by an extensive network of roads in order to access the proposed cutblocks. Large quantities of fine sediments were generated in the Prince George Forest District when similar soils and roads were exposed to erosion (Brownlee et al. 1988; Choromanski et al. 1993). The exposure of steep slopes and gullies could also increase mass wasting, adding more coarse sediments to the channels (Meehan 1991). Our detection of streambed composition changes due to spatial differences, spawning activity, and beaver activity suggest that any future logging impact would be detected. Larger streambed changes were easily quantified when coastal watersheds were logged (Scrivener and Brownlee 1989; Hartman and Scrivener 1990).

Small increases of sediment to these streams would probably not affect the sockeye salmon populations. Large quantities of fine sediments enter these streams through natural processes and are processed by freshets and the spawning fish. Sockeye salmon have also adapted to an environment rich in fine sediments by incubating their eggs during times of minimum sediment transport (Cheong et al. 1995; Scrivener and Macdonald 1996) and by producing small fry. What is unknown is how much additional sediment the populations could tolerate and what population levels are necessary to condition the incubation substrates.

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Responses of Sockeye Salmon *(Oncorhynchus nerka)* Embryos to Intragravel Incubation Environments in Selected Streams within the Stuart–Takla Watershed

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Abstract

Before impacts of forest harvesting on Pacific salmon populations can be fully understood, the natural physical and biological influences on incubation processes must be understood within interior British Columbia watersheds. The study objective was to estimate overwinter survival of sockeye salmon (*Oncorhynchus nerka*) embryos within various redd micro-environments. Egg to pre-emergent fry bioassays, in conjunction with microhabitat environmental monitoring, were implemented to define a range of natural spawning conditions and their relative contribution to fry recruitment. Four adjacent tributaries (Kynoch, Forfar, Gluskie, and Bivouac creeks) of the Stuart–Takla watershed were studied during the 1993 and 1994 broodyears. Sockeye salmon successfully spawned over a wide range of habitats. Survival rates between habitat types were not significantly different in contrast to predictions generated from optimum models. This was due to the definition of marginal habitat. Spawning adults avoided truly marginal areas with intragravel dissolved oxygen levels below 3.0 mg/L. As a result, *in situ* redd simulations showed similar intragravel conditions in both low utilization (assumed marginal) and high utilization (assumed preferred) spawning areas. Physical (i.e., hydraulic regime, bedload characteristics) and biological (i.e., mass cleaning by high densities of spawning adults) processes result in uniformly high quality gravel conditions with permeabilities, surface water interchange, and intragravel dissolved oxygen levels associated with high incubation success. Riparian-zone substrates in the study streams were characterized by large amounts of lacustrine deposits. Reduced escapement levels, or sediment inputs which exceed current bedload transport, may impact incubation environment. By spawning early in the season (July–August), the early Stuart sockeye salmon stock enjoy advanced embryological

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development prior to the onset of low water temperatures. Embryos rapidly accumulate the thermal units necessary to hatch, thereby becoming mobile in time to avoid freezing and desiccation as water-levels decline and reach seasonal minima. Embryos and alevins of the early Stuart stock can apparently tolerate temperature conditions previously considered lethal, and emerge successfully in the spring after accumulating less thermal units than any other Fraser River stock. These mechanisms are closely linked to the stream thermal regime. Riparian forestry prescriptions must be closely monitored to detect and quantify any stream temperature changes that may impact the developmental timing of incubating salmonids.

Introduction

Sockeye salmon provide one of Canada's most valuable Pacific coast fisheries (Hart 1973). The close association of salmon streams with timbered watersheds creates potential problems for fishery management (Hall and Lantz 1969; Ringler and Hall 1975; Platts et al. 1989). There has been a long history of coastal-based fish–forestry interaction research projects (FFIRP) within British Columbia (Poulin 1984; Hartman and Scrivener 1990) and the United States (Sheridan and McNeil 1968; Burns 1970; Moring 1975), but only limited study of interior watersheds (Slaney et al. 1977; Sterling 1985). Given the amount of interior forestry activity and the lack of knowledge concerning over-wintering incubation processes, there is an urgent requirement for ecological studies to guide land-use practices that are appropriate for the specific physical and biological conditions of interior watersheds.

The general objectives of this study were to define natural incubation conditions in a set of interior experimental streams which had experienced minimal anthropogenic impacts and determine responses of embryos to qualitative and quantitative differences in the various redd micro-environments in which sockeye salmon spawn. It was hypothesized that spawning salmon select incubation sites based on environmental cues to optimize egg to fry survival in northern environments.

Study Area

The Stuart River watershed represents the most northern extent of the Fraser River watershed. The early spawning Stuart River sockeye salmon stock utilizes more than 30 tributaries, primarily Takla Lake and Middle River (lat. 55°00′N, long. 125°50′W). Four adjacent tributaries (Bivouac, Gluskie, Forfar, and Kynoch creeks) were chosen for this project. Study watersheds are small streams which have no flow stabilizing lacustrine features. Twenty-six of the 33 Stuart River spawning tributaries utilized by the early spawning Stuart River stock of sockeye salmon fall into this category, representing 44% of the available spawning habitat for the stock (Langer et al. 1992). The four study streams each represent approximately 3–7% of the estimated total spawning capacity (excluding Bivouac Creek <1%), yet escapements to these four streams represent 8 to 42% of the entire early Stuart escapement (Langer et al. 1992).

Historically, the early spawning Stuart River sockeye salmon stock has never been large, and has been unusually variable (Cooper and Henry 1962; Cass 1989). Because of the geographic location and associated climate of streams used by the early Stuart spawning stock, it has been speculated that production of the early spawning Stuart River stock of sockeye salmon may be limited by environmental factors.

Materials and Methods

The main experimental approach was to: 1) map expected habitat suitability (i.e., redd distribution); 2) plant egg capsules to measure space/time variations in survival rate; and 3) monitor environmental parameters to determine their influence on the expected survival patterns.

Habitat classification was determined by observations of qualitative habitat parameters and the spatial distribution of spawners (relative densities) within each experimental reach. Two study reaches were selected in Gluskie, Forfar, and Kynoch creeks, for an initial total of six study reaches. Two study reaches on Bivouac Creek were added in 1994. Salmon observed displaying spawning behaviors had their redds marked by wooden stakes. The study reaches were partitioned, in relative terms, into low and high utilization spawning habitat. High utilization habitat was considered preferred, while low utilization habitat was considered marginal. Representative *in situ* redd simulations (2 m^2), (one marginal, one preferred) were then constructed within each study reach for an initial total of 12 redd simulations.

Bioassay techniques are designed to indicate the quality of spawning habitat by inserting eggs into the gravel in porous containers (Slaney et al. 1977; Scrivener 1988). Egg development capsules used in this study (37 mm inside diameter stainless steel cylinders punched with 2.3 mm diameter holes set at 2.0 mm centers) were modeled after Scrivener (1988). The ends were covered with snug-fitting polyethylene test caps with numerous 2.3 mm holes. A color coded wire leading from the capsule to the gravel surface marked the capsule site and assisted with retrieval. Two lengths of capsule were utilized: a standard (length = 12 cm) and a longer version for behavioral studies (length = 46 cm) (Cope 1996).

Pooled sockeye salmon gametes (4♀, 4♂) were collected from each creek. Gametes were fertilized using the wet method incorporating an isotonic sodium bicarbonate rinse. Thirty eggs, spatially separated by gravel selected for optimum qualities, were placed in each capsule. Capsules ($n_{1993} = 32$, $n_{1994} = 22$) were planted vertically in each simulated redd at a depth of 20 cm. All planting procedures were completed within 1 hour of fertilization. Fencing material was secured around each redd simulation to ensure the capsules were not disturbed by the remaining spawners. Periodic collections of developing embryos ($n_{1993} = 10$, $n_{1994} = 6$–7 capsules/redd simulation) were made in late September, December, and late April. Development rates were examined utilizing the classification system of Vernier (1969).

Environmental monitoring was implemented to determine the range of natural spawning conditions and the responses of embryos to hypothesized differences in the various redd micro-environments. Following a method developed by Terhune (1958), standpipes (10 to 26 depending on reach) were sampled before spawning (July 1–13) and during each embryo collection period. At an intragravel depth of 20 cm, dissolved oxygen, temperature, and permeability were measured. Stream temperature and dissolved oxygen were also measured adjacent to the standpipe. At each standpipe location stream depth and velocity, and visual estimates of the size of surficial streambed material were made.

Results

Redd distributions within all study reaches demonstrated spatial preferences (n = 6). Preferred spawning habitat was consistently at the tail of pools in the pool–riffle transition. Marginal habitats included: riffles; stream margins; intermittent side-channels; and portions of off-channel habitat (Fig. 1). These spatial preferences were consistent over both study years, despite significant differences in escapement and spawner densities (T. Whitehouse, Fisheries and Oceans Canada (DFO), Stock Assessment Group, West Vancouver Laboratory, West Vancouver, B.C., unpublished data).

Most incubation period mortality (80%) occurred in the first 50 days, before hatching (Fig. 2). The remaining mortality was expressed as unfertilized eggs (8%) and over-wintering processes (12%). The 1993 mean survival from fertilization to pre-emergent fry was 49%. Mean survival was 51, 50, and 46% for Gluskie, Forfar, and Kynoch creeks, respectively. An analysis of variance (ANOVA) revealed there was no significant variation in survival between creeks ($p > 0.05$), and no significant effect between streams ($p > 0.05$), stream reaches ($p > 0.05$), or habitat types ($p > 0.05$). The 1994 mean survival to pre-emergent fry was 27%. An ANOVA revealed significant variation existed in survival between streams ($p < 0.01$), but not between reaches within streams ($p > 0.05$), or habitat types within reaches ($p > 0.05$). Mean survival rate was lowest in Bivouac Creek (7%). Mean survival rates of Forfar (16%) and Gluskie (28%) creeks were intermediate, and all three creeks had lower mean survival rates than Kynoch Creek (60%).

Incubation environments were relatively invariant, with high quality incubation habitat available at all scales examined. While there were differences in the visual characteristics associated with representative marginal and preferred incubation habitats, there were no significant differences between these habitats (all reaches, creeks, and years combined) in either stream or intragravel parameters (Table 1). Although both incubation habitats contain high quality incubation conditions, there was a trend towards lower values (i.e., shallower, lower velocity, finer substrate, lower permeability, and lower intragravel dissolved oxygen) within marginal habitats (i.e., riffles; stream margins, intermittent side channels; off-channel habitat) compared to preferred spawning habitat (i.e., tail of pools in the pool–riffle transition zone).

For all streams, stream reaches, and years combined, there were no significant differences in intragravel water temperature between incubation locations, or habitat types in general (Table 2). During the mid-winter (1993) sample period, the intragravel thermal regime ($mean_{diff.} = 0.1°C$) closely paralleled the stream thermal regime. No habitat specific ground water upwelling was detectable from temperature comparisons. Generally, surface substrate composition indexes were greater than 3.4, corresponding to high quality spawning gravel with mean

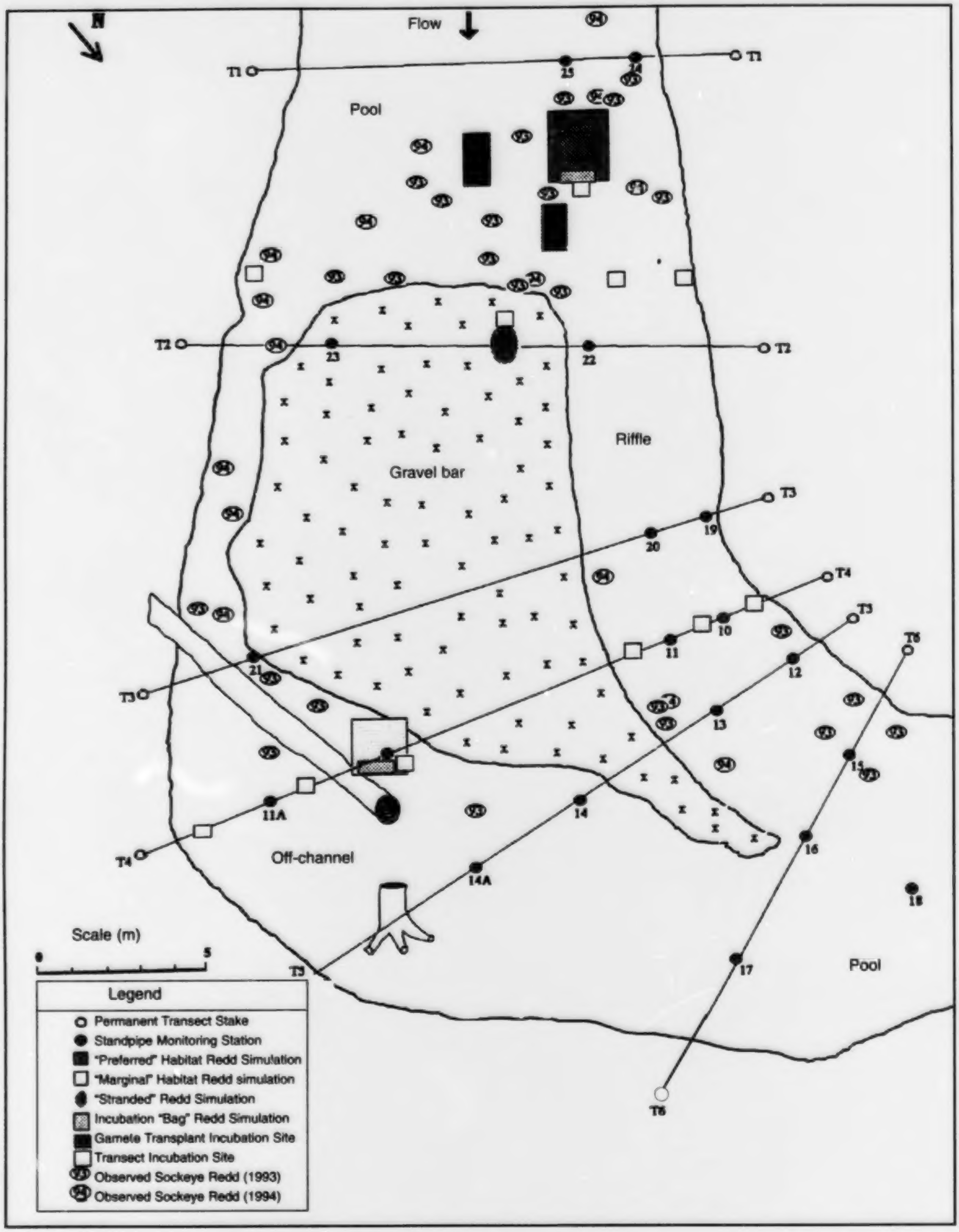

Figure 1. Kynoch Creek mid-watershed (1550 m) study reach showing standpipe monitoring stations, egg incubation sites (redd simulations), and observed redd distribution.

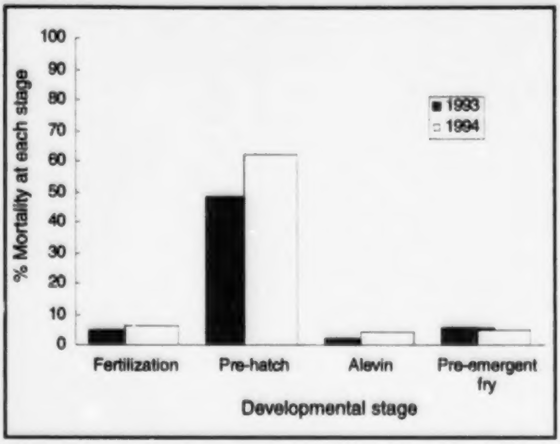

Figure 2. Percent mortality of embryos assessed at the end of each retrieval period for all study streams combined (mean +/- 2 * S.E.). Note: fertilization = 0– 2 days (August), pre-hatch = 2–50 days (late September), alevin = 50–180 days (late December to February), pre-emergent fry = 180–260 days (mid April).

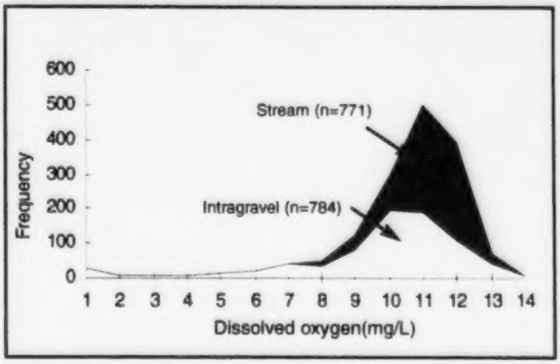

Figure 3. Frequency distribution of stream and intragravel dissolved oxygen measurements within the four study streams from the period 1992 to 1995.

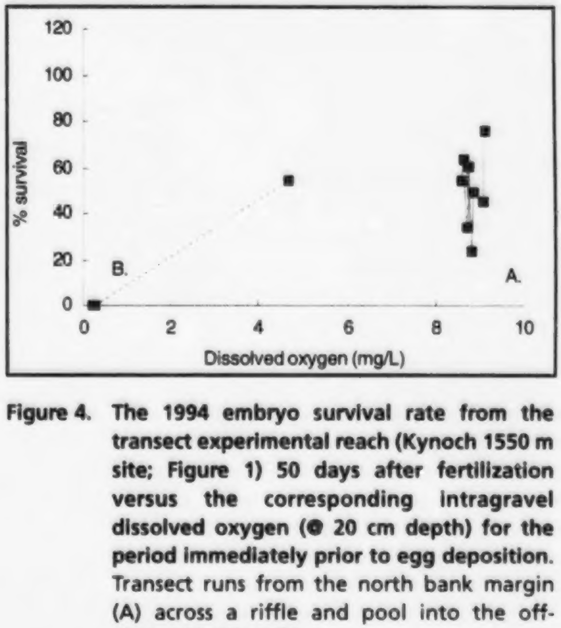

Figure 4. The 1994 embryo survival rate from the transect experimental reach (Kynoch 1550 m site; Figure 1) 50 days after fertilization versus the corresponding intragravel dissolved oxygen (@ 20 cm depth) for the period immediately prior to egg deposition. Transect runs from the north bank margin (A) across a riffle and pool into the off-channel habitat (B).

particle sizes of (2–64 mm). Only off-channel habitats had a significant proportion of silt and sands. Upper locations were coarser due to increasing gradients and water velocities ($p < 0.05$). Broodyear differences were attributed to water impoundment by beavers during the winter of 1993–1994 in two of the six study reaches. This resulted in surface deposition of fine sediments. All habitats (Table 2) and capsule incubation locations (Table 1) contained mean permeabilities greater than 19 mL/s. Ninety percent of all samples (n = 784) of stream and intragravel dissolved oxygen were greater than 6.0 mg/L (Fig. 3). Across habitat types, only off-channel habitat had lower mean intragravel dissolved oxygen ($p < 0.05$; Table 2).

Multiple regression analysis of embryo survival rates on physical variables measured at incubation location standpipes (stream temperature, dissolved oxygen, velocity, depth, surficial substrate index, and intragravel temperature, dissolved oxygen, permeability) failed to detect any significant correlation predictions ($p > 0.05$, $r^2 = 0.17$, $n = 53$). There was a weak relationship between intragravel dissolved oxygen and survival rate ($p = 0.06$, $r^2 = 0.34$, $n = 43$). Transect data (n = 11) suggest intragravel dissolved oxygen does not effect survival rate until levels drop below 4.0 mg/L (Fig. 4). Furthermore, spawning sockeye salmon did not utilize habitat with less than 3.0 mg/L intragravel dissolved oxygen. Embryos placed in these locations did not survive, and this relationship is probably not well described by linear regression models.

The thermal regime in Forfar Creek was representative of the annual pattern for natal streams used by the early Stuart River sockeye salmon stock. Daily stream temperatures rise from 4–10°C during June, vary from 8–16°C during summer, and drop rapidly in October and remain at 0.0–0.5°C throughout winter (Fig. 5). Comparisons of the estimated

Table 1. Summary of stream and intragravel physical parameters for preferred and marginal redd simulations (all locations, creeks, and years combined)

Variable	Capsule incubation site	
	Preferred	Marginal
Velocity (m/s)	0.19	0.13
S.E.	0.02	0.04
Range	0.00–0.45	0.0–0.71
Sample size	22	21
Depth (cm)	29.1	20.7
S.E.	4	3.7
Range	2.0–74.0	1.0–57.0
Sample size	24	23
Surface substrate	3.41	3.06
S.E.	0.22	0.3
Range	1.0–4.4	1.0–5.0
Sample size	18	18
Stream temperature (°C)	3.58	3.4
S.E.	0.65	0.63
Range	–0.1–9.4	–0.1–8.8
Sample size	25	24
Intragravel Temperature (°C)	3.6	3.42
S.E.	0.65	0.62
Range	–0.1–9.4	0.0–8.5
Sample size	25	24
Stream dissolved oxygen (mg/L)	11.68	11.17
S.E.	0.17	0.29
Range	10.3–13.1	7.3–12.9
Sample size	25	23
Intragravel Dissolved oxygen (mg/L)	10.76	9.99
S.E.	0.22	0.4
Range	8.2–12.4	4.9–12.0
Sample size	25	24
Permeability (mL/s)	21.7	19.3
S.E.	2.8	3.7
Range	8.4–50.0	0.0–48.6
Sample size	16	16

S.E. = Standard error.

Note: ANOVA ($p < 0.05$).

hatching interval (September 27–October 29) and data from stream thermographs suggest alevin hatching coincides with the approximate fall freeze-up period. Spawning coincides with maximum annual stream temperatures and incubating embryos rapidly accumulate thermal units early in development. As a result, 67% of the thermal units are accumulated within the first 19% of the incubation period. The range of thermal units at this point (October 29) was 372–436 accumulated thermal units (ATU). By early December, during the onset of minimum temperatures and flows, 100% of embryos had reached the alevin stage (stage

Table 2. Summary of stream and intragravel physical parameters by habitat sampling unit (margin, thalweg, pool, and off-channel) for all seasons, creeks, and locations combined

	Habitat unit				ANOVA
Variable	Margin	Thalweg	Pool	Off-channel	$p < 0.05$
Velocity (m/s)	0.29	0.43	0.14	0.08	***
S.E.	0.02	0.02	0.01	0.01	
Range	0.01–1.06	0.00–1.45	0.00–0.58	0.00–0.59	
Sample size	166	187	90	100	
Duncan*	B	A	C	D	
Depth (cm)	18.9	27.5	41.8	25.3	***
S.E.	1	1.1	1.9	2	
Range	3.0–64.0	2.0–100.0	2.0–84.0	1.0–132.0	
Sample size	168	190	92	98	
Duncan	C	B	A	B	
Surface					
Substrate	3.57	3.99	3.38	2.84	***
S.E.	0.07	0.05	0.16	0.11	
Range	1.7–5.1	2.0–5.0	1.0–5.2	1.0–5.0	
Sample size	164	184	73	82	
Duncan	B	A	B	C	
Stream					
Temperature (°C)	6.7	6.6	5.9	5.8	
S.E.	0.3	0.3	0.4	0.4	
Range	−0.1–12.7	−0.1–12.7	−0.1–12.7	−0.2–12.9	
Sample size	171	194	93	98	
Intragravel					
Temperature (°C)	6.7	6.6	6	5.7	
S.E.	0.2	0.3	0.4	0.4	
Range	−0.1–12.8	0–12.8	−0.1–12.7	−0.1–12.8	
Sample size	172	193	93	98	
Stream					
Dissolved oxygen (mg/L)	10.9	11	11.1	10.4	
S.E.	0.1	0.1	0.1	0.2	
Range	9.4–13.3	9.4–13.2	9.1–12.8	3.6–13.0	***
Sample size	171	194	93	98	
Duncan	A	A	A	B	
Intragravel					
Dissolved oxygen (mg/L)	9.5	9.8	9.9	6.6	
S.E.	0.17	0.15	0.2	0.38	
Range	0.3–12.8	0.3–12.9	0.3–12.4	0.2–12.0	***
Sample size	172	193	93	98	
Duncan	A	A	A	B	
Permeability (mL/s)	25.4	25.9	24.9	19.9	
S.E. 1.5	1.5	2.5	1.8		
Range	3–125	4–129	6–120	0–82	
Sample size	160	181	60	78	

S.E. = Standard error.

*** Indicates significant difference at the $p < 0.05$ level.

* Duncan refers to results from post hoc Duncans multiple range test.

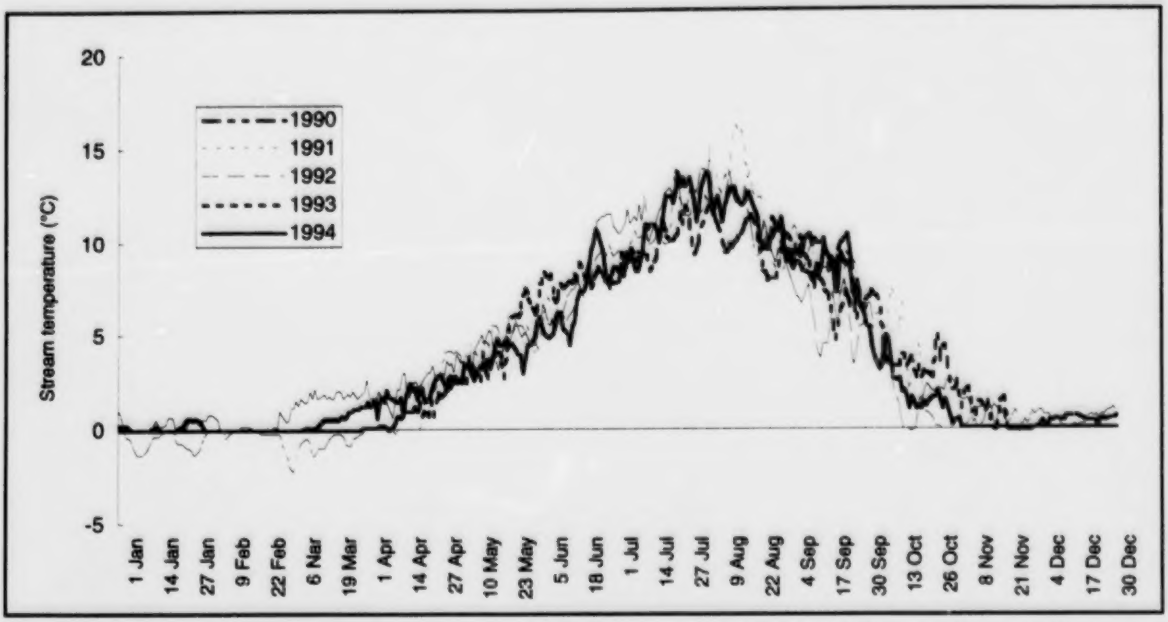

Figure 5. Daily maximum temperatures in Forfar Creek for the period 1990 to 1994 (B. Anderson, Fisheries and Oceans Canada, Pacific Biological Station, Nanaimo, B.C., unpublished data).

31–35). Coinciding with rising stream temperatures in spring, fry emergence occurs principally from mid-April to mid-May after exposure to 600–800 thermal units (mean = 260 days).

There was no difference in intragravel dissolved oxygen ($p > 0.05$) between stranded (marginal sites subject to exposure) and control (preferred) *in situ* redd simulations. Samples taken at a depth of 40 cm (11.3 mg/L, 8.6 mg/L) indicate hospitable rearing environments at depths below the mean redd depth of 20 cm. No control sites were subject to desiccation or freezing (water level above surficial substrate) while 100% of the stranding simulations were influenced by declining water levels. The vertical distribution of alevins within behavior capsules in relation to depth of freezing confirms alevins are responding directly to this stimulus rather than to some other factor that might promote better survival by moving deeper into the gravel (Fig. 6).

Discussion

Preferred habitat was the downstream ends of pools at the pool–riffle interface, previously noted in many studies (Hoopes 1972; Tautz and Groot 1975; Bjornn and Reiser 1991). Marginal habitats which were utilized to a lesser degree included riffles, stream margins, intermittent side channels, and portions of the off-channel habitat. Sockeye salmon successfully spawned over this wide range of habitats. This range likely has upper and lower limits beyond which fish were unwilling to spawn as observed from the lack of spawning in habitat where intragravel dissolved oxygen levels were below 3.0 mg/L. Based on the lineal distribution of spawning adults and their corresponding micro-habitat parameters, few locations in the lower and mid-watershed reaches contained these zones of exclusion (Tschaplinski 1994).

Spawning sockeye salmon select spawning habitat to optimize egg to fry survival, and as predicted by the gradation in habitat model (Hilborn and Walters 1992), once local densities reach certain limits the remaining spawners move on to colonize less densely populated reaches or streams rather than spawn in unsuitable habitat, or at dangerously high densities on a local scale. Spawner escapement estimates for the study streams were very different between the 2 years of study (T. Whitehouse, DFO, Stock Assessment Group, West Vancouver Laboratory, West Vancouver, B.C., unpublished data). Based on estimates for usable spawning area optimum escapement levels were exceeded in 1993 (density = 3.46 spawners/m^2), while 1994 was well below this level (density = 0.30 spawners/m^2). During the high density broodyear, spawner abundances were dramatically higher at all locations, and distributions extended 1.2–2.0 km further upstream to where

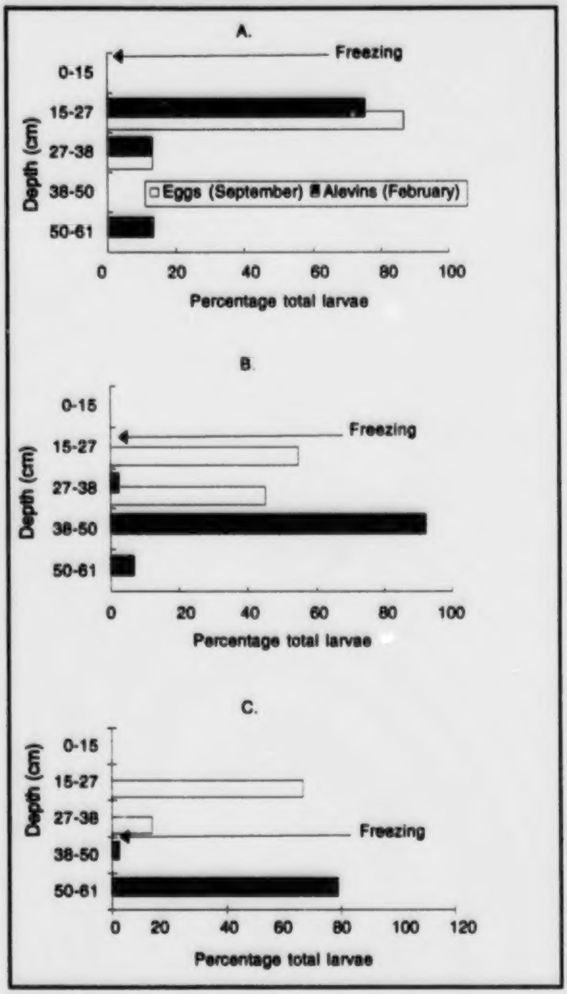

Figure 6. Ingravel pattern of changes in vertical distribution of sockeye alevins in relation to depositional depth (eggs in late September) and depth of intragravel freezing. A 2-cm, 15-cm, and 36-cm depth of freezing within substrate for diagram A, B, and C, respectively.

obstructions blocked upstream movement (P. Tschaplinski, Ministry of Forests, Research Branch, Victoria, B.C., personal communication). However, the poor quality incubation habitat (i.e., dissolved oxygen less than 3.0 mg/L) were generally not utilized for spawning even in this situation. Instead, a larger scale spatial re-distribution of spawners occurred. This escapement re-distribution in dominant cycle years was a previously noted phenomenon of the early Stuart River stock (Langer et al. 1992).

The survival rate to pre-emergent fry reflects the overall rigors endured by a given population of developing eggs and alevins and is a consequence of the severity of the environmental conditions and the adaptability of the fry (Koski 1966). The capability of the spawning grounds in these northern interior streams to sustain eggs and alevins from spawning to pre-emergence was inferred from the results using the perforated incubation capsules. These bioassays were designed to indicate the quality of spawning habitat for the embryo and alevin stage of incubation, but they do not provide an assessment of the critical stage of alevin to emergent fry.

The primary limitation of concern was over-wintering egg mortality due to freezing and dewatering. Stream temperatures declined to mid-winter lows of 0°C for several months, and up to 80% declines in water level were observed after the spawning period (Scrivener and Anderson 1994). However, during both years of study, the majority of mortality (80%) occurred before the onset of winter conditions, and only 12% of embryo mortality occurred from 10 October to 15 April (Fig. 2). These results conform to patterns derived for *Oncorhynchus* spp. from coastal and laboratory systems (Murray and McPhail 1988; Scrivener 1988; Beacham and Murray 1989; Cowan 1991). This suggests over-winter mortality due to freezing and desiccation was not determining fry production as originally hypothesized.

Observed survival rates were facilitated by spawning adults selecting incubation microhabitat to optimize egg to fry survival and a number of general mechanisms which would optimize incubation success in northern environments. These mechanisms included the early time of spawning, thermal tolerance, development rate, alevin behavioral mechanisms, and habitat modifications by spawning adults.

Mean survival rates to pre-emergent fry of 49% and 28% for 1993 and 1994, respectively, are indicative of a high quality incubation habitat. Productivity, related to density-independent mechanisms, for Kynoch, Forfar, and Gluskie creeks appears to be very high. Egg to pre-emergent fry survival rates compared favorably to pre-emergent fry survival rates for similar studies (*Oncorhynchus* spp.) in coastal systems (Scrivener 1988; Cowan 1991). Further, the early Stuart River stock does not exhibit lower overall recruitment rates per spawner than other Fraser River stocks (Walters and Staley 1987).

In contrast to predictions generated from optimality models (Fretwell and Lucas 1970; MacCall 1990, Hilborn and Walters 1992), survival rates between assumed preferred and marginal habitats were not significantly different. This was due to the perception and definition of marginal habitat. Truly marginal areas (i.e., less than 3.0 mg/L dissolved oxygen) were avoided by spawning adults. The result was low utilization (i.e., assumed marginal) and high utilization (i.e., assumed preferred) *in situ* redd simulations with similar intragravel conditions.

While numerous studies have led to the consensus that low dissolved oxygen and reduced water exchange increase embryo mortality, variation due to other factors often obscures this relationship in natural systems, such that survival often appears independent of intragravel dissolved oxygen (Koski 1966; Chapman 1988). Hansen (1975) found streambed areas with low dissolved oxygen (less than 3.0 mg/L) are not used for spawning. As 90% of the recorded dissolved oxygen values within this study were greater than 6.0 mg/L, and those areas less than 3.0 mg/L were not utilized for spawning, it was not surprising to find a significant relationship did not exist between intragravel dissolved oxygen and survival.

Survival of salmonid embryos has been related to substrate composition in many experiments and field studies (see Chapman 1988 for review). High intragravel dissolved oxygen and survival rates have often been attributed to high permeabilities. Permeability does not affect survival directly but is a measure of the ability of the gravel in the redd to allow for a sufficient supply of water and dissolved oxygen to the embryos and fry. The spawning female can alter grain size and porosity of gravel to ensure that incubation begins with an adequate flow of oxygenated water (Chapman 1988). The female vigorously removes fines and small gravel to form the egg pocket; when completed the redd will contain less fine silt and sand than the surrounding substrate (Chapman 1988). Substrate permeabilities within the study streams closely agree with values documented by Chapman (1988) for heavy spawning gravel beds of optimal survival conditions. Concurrent studies by other researchers demonstrate that particles less than 0.3 mm in diameter were rare (1–1.6%) and interstitial spaces in the gravel would remain clear permitting water exchange and movement of alevins (Scrivener 1994). The lowest mean permeabilities at the completion of the incubation period were greater than 19 mL/s (Cope 1996). Koski (1966) reported no detectable affect on survival above this threshold. Egg to fry survival rates of greater than 30% are expected from such gravels (Lotspeich and Everest 1981).

Large annual escapements of spawning Pacific salmon probably engender a mass cleaning and help maintain high quality spawning habitat (Everest et al. 1987). Using independent methods, Scrivener and Anderson (1994) and Gottesfeld (1994), estimated spawning salmon within the study streams accounted for 25–50% of the annual movement of bed material. It is possible that over a number of years the geomorphic work done by spawning sockeye is comparable to, or even greater than, that performed by floods (Gottesfeld 1994).

For fall spawners, embryos must reach some critical stage of development before the water becomes to cold (Combs and Burrows 1957; Combs 1965). Results suggest that pink salmon (*Oncorhynchus gorbuscha*) and chinook salmon (*Oncorhynchus tshawytscha*) embryos could tolerate long periods of low temperatures if initial temperatures are above 6.0°C and embryogenesis had proceeded to a critical developmental stage. The early spawning Stuart River sockeye salmon stock spawns four weeks earlier than any other Fraser River sockeye salmon stock (Killick 1955; Brannon 1987). By spawning early, offspring of early spawning Stuart River adults experience a rapid early development phase and are undergoing the critical hatching phase as freeze-up descends upon the region.

The observed downward migratory response of sockeye salmon alevins to freezing (Fig. 6) is an adaptation to the incubation environment which allows them to avoid/survive the apparent harshness of central interior streams. Interstitial migratory behavior in response to substrate freezing was also demonstrated in laboratory studies utilizing early Stuart River broodstock. This response was not observed in offspring of coastal sockeye broodstock, suggesting there is a genetic determinant of behavior, which the coastal alevins lacked (C. Scrivener, DFO, Pacific Biological Station (PBS), Nanaimo, B.C., personal communication).

The trade-off against the early spawning strategy is the effect of unusually stressful migration conditions on the quality and viability of the gametes. Evidence of this trade-off was obtained in 1994 when egg survival rates were very low for spawners that arrived late (Cope 1996) and coincidentally, had suffered severe thermal stress during migration (Clarke et al. 1995). Implications for stock management are the general assumption of classical recruitment models that variation in recruits/spawner must be due to

variations in environmental conditions experienced by the recruits, and not their parents. The apparent adaptation in developmental biology of the early spawning Stuart River stock and the specific timing to the seasonal variation in temperature and hydraulic regime would suggest monitoring these variables in association with forestry prescriptions should be a priority of future research. As riparian zone substrates within the study area were characterized by large amounts of lacustrine deposits (Sanborn 1994), post-logging increases in the delivery of fine sediments must not surpass the ability of physical (hydraulic regime, bedload characteristics) and biological (mass cleaning by high densities of spawning adults) processes to maintain the current gravel quality.

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The Abundance, Distribution, and Habitat Preferences of Adult Sockeye Salmon in Streams Tributary to Takla Lake and the Middle River of Central British Columbia

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Abstract

The abundance and longitudinal distribution of adult sockeye salmon (*Oncorhynchus nerka*) returning to spawn in Gluskie, Forfar, and O'Ne-ell creeks have been determined annually since 1992 to identify spawner habitat use and preferences. Spawner distribution relative to key physical habitat features is being determined by using aerial and ground-based habitat inventory techniques combined with both total-area enumerations and strip-counts for spawners. Habitat descriptions are obtained at three scales: (1) macrohabitat (stream reach), (2) mesohabitat (features such as pool, riffle, and glide types including channel structure), and (3) microhabitat (substrate texture, water depth, velocity, and spatial indices). The overall goal in this project is to determine how forestry-associated changes in the thermal, hydrologic, and geomorphologic regimes in the selected study watersheds affect variations in (a) the structure, stability, and distribution of fish habitats, (b) sockeye salmon spawner distribution and habitat use, and ultimately, (c) egg-to-fry survival.

Introduction

The Stuart–Takla Fisheries–Forestry Interaction Project is a multi-agency and multidisciplinary investigation of the effects of modern forest-harvesting practices on stream ecosystems, salmonid populations, and fish habitats in the interior of British Columbia. The relationships between forestry activities and the productivity of aquatic ecosystems in the central interior of B.C. are not well known because relatively little research has been undertaken in this region. Research programs in coastal watersheds such as Carnation Creek (Vancouver Island) and on the Queen Charlotte Islands have provided much of the information upon which the provisions of the new (1995) B.C. Forest Practices Code are based (*see* Hogan and Tschaplinski 1996; Hartman and Scrivener 1990). Because of regional differences in climate, topography, hydrology, and soils, the results of over 25 years of research conducted in coastal B.C. systems is frequently not applicable in the interior. Thus, Forest Practices Code provisions are frequently adapted for interior ecosystems from extrapolations of the knowledge gained on the coast. Operational testing and evaluation of these provisions will aid in distinguishing coastal watersheds from those located in the central interior of B.C.

The Stuart–Takla study was initiated in 1990; it incorporates several watersheds and is the first of its kind in the B.C. interior. The Takla Lake drainage is located in the northernmost part of the Fraser River

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basin (Macdonald et al. 1992). The area contains valuable forestry, fisheries, and recreational resources. It is especially important for the production of sockeye salmon[1] (*Oncorhynchus nerka*). The Fraser basin is one of the most important salmon producing systems in North America. On average, about 25% of the annual production of sockeye from the Fraser River originates in the Stuart–Takla drainage. Research on fisheries–forestry interactions is vital to ensure that these prime salmonid spawning and rearing habitats are protected in the future and to aid in making ecologically sound decisions about integrated resource management.

Accordingly, the overall objectives of this long-term research project are to: (1) provide an understanding of the physical and biological processes operating within several watersheds in the central interior region of B.C.; (2) determine how forest-harvesting practices conducted according to the Forest Practices Code change these processes; and (3) apply the results to refine the current harvesting and resource protection provisions for interior forests within the new Forest Practices Code.

The project is based on four streams tributary to Takla Lake and the Middle River which serves as the lake's outlet. Researchers began collecting pre-harvest baseline data in 1990 from three unlogged watersheds, Gluskie, Forfar, and O'Ne-ell (Kynoch) creeks, and from Bivouac Creek which has been partially logged. Each stream contains populations of bull trout (*Salvelinus confluentus*), rainbow trout (*Oncorhynchus mykiss*), "kokanee" (*Oncorhynchus nerka*), and especially sockeye salmon. Together, these streams sometimes receive over 50% of the total number of adult sockeye returning to spawn in the Takla Lake drainage in summer (Macdonald et al. 1992).

Logging conducted under the provisions of the Forest Practices Code do occur in parts of the study area. The study will feature experimental controls in both area and time. For example, the Forfar Creek watershed will remain unlogged for the duration of the project in order to evaluate the effects of different harvest options conducted in the other sites. A minimum of 4 years of pre-harvest data will be collected from each of the remaining watersheds. The project design and component studies are detailed in other presentations within these proceedings and published elsewhere (Macdonald et al. 1992; Macdonald 1994).

This component of the overall program focuses on adult sockeye spawning habitat ecology. Sockeye use the study streams primarily for spawning and egg incubation (Tschaplinski 1994). Spawners arrive at the mouths of the streams in mid-to-late July after migrating about 1300 km from the mouth of the Fraser River in roughly 18 days. Spawning is essentially concluded by about 18 August. Eggs and alevins develop and remain in the streambed until the following spring. The majority of sockeye fry emerge from the streambed during April and May and almost immediately emigrate downstream to rear for about 1 year in the lake system. Because most fry spend relatively little time in their natal streams, the direct effects of forest harvesting upon sockeye fry production from the study streams will likely be upon (a) adult spawners and the quantity, quality, and stability of stream spawning habitat, and consequently (b) the survival and development of eggs and embryos in response to streambed conditions.

Accordingly, the objectives of this project component are to: (1) determine the abundance and spatial distribution of adult sockeye salmon in Gluskie, Forfar, and O'Ne-ell creeks; (2) determine spawner densities and habitat preferences throughout each stream by (a) habitat type, and (b) microhabitat characteristics including water depths, stream velocities, spatial indices (stream widths and volumes), and substrate texture; (3) interpret spawner site-selection (densities) relative to variations in egg-to-fry survival and fry production determined by other project investigators; and (4) monitor for post-harvest changes in habitat type, stream channel structure and stability, and spatial distribution of adult sockeye.

[1] For brevity, sockeye salmon are frequently referred to as sockeye in this paper.

Methods

Details of all stream survey methods noted here are given by Tschaplinski (1994), and Tschaplinski and Hyatt (1990, 1991). The following summary describes work in progress. Preharvest data on adult distribution and habitat preferences have been collected since 1992.

Sockeye Abundance

It is necessary to accurately enumerate the numbers of adult sockeye entering each stream because the effects of forest harvesting on sockeye production will ultimately be evaluated from long-term trends in egg-to-fry survival. These survivals reflect the study streams capacity to produce fry from a given number of spawners, and are determined by comparing the number of fry migrating from the streams each spring with estimates from the total number of eggs deposited in the previous summer. Egg deposition is calculated from the seasonal abundance of spawners, the sex composition of the spawning run, and estimates of female fecundity.

Because accurate estimates of spawner abundance are critical components of this project, several methods have been simultaneously employed to enumerate adult sockeye on the spawning grounds. The accuracy and precision of each method varies to different degrees according to variations in environmental conditions and other factors (Tschaplinski and Hyatt 1990, 1991). For example, flash floods may damage counting fences and allow migrating fish to escape upstream without being enumerated. By employing different techniques, acceptable estimates of abundance are obtained each year by at least one method.

Staff from Fisheries and Oceans Canada (DFO) obtained daily counts of sockeye by sex at temporary fences located near the mouths of each study stream (Smith 1994; Macdonald et al. 1992). These fence counts were supplemented every second day by visual counts of both live and dead spawners obtained by surveyors walking the entire length of each stream occupied by sockeye (Smith 1994).

Alternate estimates of sockeye abundance were produced in this study to further supplement those obtained by DFO. These estimates include those determined by using (1) Petersen mark–recapture techniques, and (2) the area-under-the-curve method with an estimate of stream-residence time for spawners (see Tschaplinski and Hyatt 1990, 1991). Both techniques required the application of tags to large samples of fish when they entered the study streams. Fish were tagged in two daily sessions for each stream, once at the beginning and again near the chronological mid-point of the seasonal spawning migration. In each stream, up to 500 sockeye or more were tagged per day with brightly colored plastic discs of 25-mm diameter (Tschaplinski and Hyatt 1990, 1991). A unique color of tag was used for each stream on each tagging date.

During each tagging session, sockeye were captured with seine nets downstream of the counting fence. One set of two discs was fixed to the anterior base of the dorsal fin of each fish (Tschaplinski and Hyatt 1990, 1991). Tagged fish were then released upstream of the fence so that they could continue their migration to their spawning sites. A gate in the fence was opened as needed to permit the passage of any fish attempting to return downstream.

Both tagged and untagged sockeye were enumerated throughout the spawning grounds of each stream at intervals spaced at 6–7 days during the spawning season. The first fish survey was conducted 3 days after the tags were first applied in order to allow tagged fish sufficient time to disperse throughout the spawning grounds. These surveys were conducted not only to determine spawner abundance, but also to determine (1) migration patterns and distribution of spawners arriving at different times of the migration period, (2) preferred sockeye habitat prior to forest harvesting, and (3) competition for space between early-arriving and later-arriving spawners.

Sockeye Distribution and Habitat Preferences

Two methods have been employed to determine adult sockeye migration patterns, distribution, and habitat preferences. Both techniques also generated additional estimates of population abundance. One method is the strip-count procedure (Tschaplinski and Hyatt 1990, 1991). This method involves establishing counting stations at regular intervals along the entire length of each stream occupied by adult sockeye. Stations are spaced at 30-m intervals in the present study. The 30-m interval has been found to effectively cover the full range of repeating habitat units (e.g., riffles, pools, glides) in streams 8–10 km long where channel widths in most areas vary, usually between about 8 and 30 m (Tschaplinski and Hyatt 1990, 1991).

During each fish survey, observers walked the entire length of the spawning grounds but counted fish only at the strip-count stations. At each station, all fish visible (both live spawners and carcasses) in a 1-m wide cross-section or "strip" of the stream

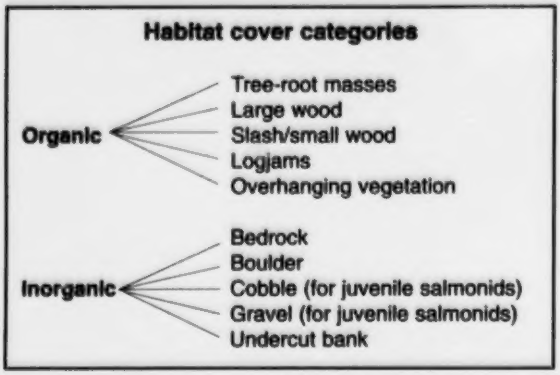

Figure 1. Categories used to classify the habitat occurring at each strip-count station in the study streams.

were counted. Thus, these counts are a sample of the total population. Each count is also a site-specific measure of the density of spawners in terms of numbers per lineal metre of stream. An estimate of the total population was generated when densities at the 1-m wide strips were expanded over the entire length of the stream inhabited by adult fish.

At the same time that strip counts were performed, other observers counted all live spawners and carcasses visible throughout the stream, and summed their counts for each 30-m interval between the strip-count stations. The distribution of all spawners in the stream was thus determined by 30-m interval by this second procedure. This total-population count was performed to evaluate how accurately the population subsample derived from the strip counts reflected the actual longitudinal distribution of sockeye on the spawning grounds.

The accuracy of both population assessment methods was determined by comparing them to sockeye abundance estimates based upon fence counts. Except when fish fences are damaged or submerged by floodwater, the most accurate estimates of adult salmon abundance entering streams are always generated from counts of migrants made at fences located at stream mouths. In the present study, the numbers of live sockeye on the spawning grounds on any given day was determined from the cumulative seasonal count of sockeye passing through the fish fences adjusted by (1) adding the numbers of sockeye distributed downstream of the fences at the mouths of the streams, and (2) subtracting the cumulative numbers of sockeye carcasses counted upstream of the fences up to the day in question. Fence counts that supplement the data are termed "adjusted fence counts" in this report, and serve as our primary reference for sockeye population abundance on any given date. The numbers of sockeye occurring downstream of the fish fences, and the cumulative carcass counts were provided by stream surveys conducted by DFO crews.

Macrohabitat-Level Assessment

Sockeye habitat was determined at three spatial scales. The broadest scale employed was at stream reach level (British Columbia Ministry of Forests and British Columbia Ministry of Environment 1995). At this macrohabitat level, the longitudinal distribution of adult sockeye was mapped from the mouth of the stream to the their upstream limit. A reach is a relatively homogenous section of a stream having a sequence of repeating structural characteristics (or processes) and fish habitat types (British Columbia Ministry of Forests and British Columbia Ministry of Environment 1995). The key physical factors used to determine reaches in the field are channel pattern, channel confinement, gradient, and predominant streambed and bank materials. Stream reaches generally show uniformity in these characteristics and in discharge (British Columbia Ministry of Forests and British Columbia Ministry of Environment 1995).

Reach boundaries were defined to occur at (1) significant changes in stream channel form or confinement, such as from a single channel to braided, multiple channels, or at the change from a wide floodplain to a confined canyon; (2) significant changes in gradient; (3) significant changes in streambed or streambank materials; and (4) potential barriers to fish distribution such as major waterfalls. The physical characteristics that differentiate stream reaches also define the fish habitats they contain, and determine their ability to support fish populations.

Mesohabitat-Level Assessment

Spawner preferences at the mesohabitat or intermediate scale were determined by using logistic regressions (McCullagh and Nelder 1983) to compare spawner densities observed at strip-count stations with the frequency of habitat types occurring across all strips. The habitat at each strip was classified into standard categories for pool and fast-water habitats (Fig. 1) by using a scheme adapted from those of Bisson et al. (1982) and Sullivan (1986). The approximate proportions of substrate types were also estimated visually by referring to standard size categories (Fig. 2). Also recorded were features such as cover afforded by pieces of large woody debris (enumerated pieces of LWD; *see* Hogan and Chatwin

Substrate proportions	
Boulder	> 256 mm
Cobble	> 64–256 mm
Gravels	6–64 mm
"Pea" gravel	> 2–5 mm
Sand	> 1–2 mm
Fines	< 1 mm
Fine organics	

Figure 2. Categories used to describe cover features and substrate surface composition occurring at each strip-count station. See text for methods of cover quantification. At most strips, substrate types were ranked in order of approximate surface area of streambed covered. At selected strips, actual proportions of each size class were determined from photographs.

1994), riparian canopy vegetation (approximate percent of stream shaded), and presence of streambed boulders (Fig. 2).

Habitat inventories were obtained for all strips prior to the spawning season because spawning sockeye can modify some habitat characteristics, especially the particle size composition of the streambed (Scrivener 1994). Prespawning habitat inventories thus permitted the determination of habitat preferences from variations in fish density recorded when spawners were on site. Habitat inventories were repeated after the spawning season to evaluate the effects of adult sockeye on streambed sediment textures. Habitat classifications obtained at each strip during ground-based surveys were supplemented annually with a continuous sequence of stereo aerial photographs (200-mm or 70-mm lenses) taken of the entire stream channel (Hogan 1994).

Microhabitat-Level Assessment

Immediately after strip-count surveys were performed, a subset of strips was selected for each stream by random sampling stratified by fish density. The observed range of spawner densities was thus included within the sample. The sample usually varied between 9 and 13 stations and has represented between about 12 and 20% of all strip-count stations located in the lowermost reach of each stream used by sockeye. At each cross-stream strip, measurements of microhabitat variables were made including stream velocity, streambed surface texture (particle size composition), water depth, and available habitat space (stream wetted width and water volume). Linear correlations were later employed in the laboratory to determine the statistical associations between spawner density and these microhabitat characteristics.

The wetted width of the stream was measured with a metre tape. Water depths, velocities, and 35-mm photographs of the streambed were taken along the cross-stream transect for each strip at either 0.5- or 1.0-m intervals depending upon stream width. One-metre intervals were used when wetted width was ≥ 10 m. Water depths were measured with metre sticks, and an electronic current meter was used to determine water velocities. Velocity readings were taken at 20 and 80% of the total depth at each 0.5- or 1.0-m interval. For transect locations where the stream was <10 cm deep, a single velocity reading was taken at 50% depth. Mean velocities for each strip were based usually upon 10 to 32 individual readings. Mean depths were based usually upon seven to 20 measurements. Where very shallow water occurred along the banks (i.e., 1–3 cm deep), measurements made within 0.5 m of the bank were not used for subsequent determinations of average depth or velocity. Substrate photographs were taken with a combination of single-lens-reflex land cameras fitted with polarizing filters or similarly-fitted underwater cameras. Photographs were later analyzed in the laboratory (see Tschaplinski 1994 for further details of all methods).

Results and Discussion

Annual trends in adult sockeye abundance and distribution were consistent across all study streams. Therefore, variations in abundance and distribution within and among years are frequently illustrated in this report by using O'Ne-ell Creek as a typical example.

Sockeye Abundance and Comparisons of Assessment Methods

The numbers of adult sockeye returning to each study stream, and the corresponding densities of fish on the spawning grounds have fluctuated widely among years (Table 1). For example, based upon daily fence counts (adjusted), the numbers of live spawners enumerated in early August was over sixfold higher in O'Ne-ell Creek in 1995 compared with 1994 (Table 2). Because fish fences can provide absolute counts of migrating salmon entering streams, the interannual variation in the numbers of sockeye returning to the project study streams has been accurately tracked. However, expansions of

Table 1. Mean densities (numbers per lineal metre) of sockeye spawners in the lowermost reaches of the Stuart–Takla study streams from 1992 to 1995. Densities were averaged across all strips in the lower about 1.3, 2.7, and 3 km of Gluskie, Forfar, and O'Ne-ell creeks, respectively (study section 1). Note that section-1 densities do not vary proportionally with population levels because many spawners move upstream to occupy steeper-gradient reaches in each stream when densities exceed about 4.5 fish/lineal m.

Year	Stream	Seasonal (s) or strip-count (*) population abundance for entire stream	Section 1 mean density (no./lin. m) on survey date	Standard deviation (+/−)	Comments
1992	Gluskie	3 749 (s)	1.5	1.7	Densities based on 1860 live fish on survey date.
	Forfar	12 904 (s)	1.7	2.0	Densities based on 4240 live fish on survey date.
	O'Ne-ell	11 002 (s)	1.5	3.6	Densities based on 3870 live fish on survey date.
1993	Gluskie	19 556 *	10.0	5.1	Densities based on strip-counts near peak of
	Forfar	15 990 *	4.6	2.7	spawning in each stream, Significant portions of
	O'Ne-ell	26 225 *	6.5	3.6	the total population inhabited section 2 upstream.
1994	Gluskie	3 919 (s)	2.7	4.5	Almost entire seasonal migration for each stream
	Forfar	4 902 (s)	2.9	3.4	was present and alive on spawning grounds on
	O'Ne-ell	4 371 (s)	1.8	1.7	date of survey.
1995 Early season	Gluskie	13 689	8.4	8.8	Numbers of live fish for each survey period in
	Forfar	14 859	6.8	2.6	1995 were determined from daily fence counts
	O'Ne-ell	23 857	6.1	4.3	plus DFO stream surveys.
1995 Near peak of spawning	Gluskie	9 510	7.4	8.8	
	Forfar	10 804	4.4	2.6	
	O'Ne-ell	18 063	6.1	4.3	

strip counts have provided estimates of population abundance that closely approximated those generated from adjusted fence counts for all streams in most years. In 1995, strip-count expansions were within 16.3 and 12.9% of the equivalent fence counts for the same survey date for early-season and mid-season population assessments respectively (Table 2). In 1992, the numbers of live sockeye estimated from expansions of strip counts were within 5.6–13.3% of estimates based upon adjusted fence counts (Tschaplinski 1994).

The accuracy of fence counts decreased in 1993 because floods damaged the fish fences and allowed many spawners to migrate upstream without being enumerated. Given the relatively consistent correspondence between strip-count expansions and adjusted fence counts in our study streams, strip counts provided useful alternate estimates of population abundance for specific survey dates in 1993 (Table 1).

To date, strip counts have shown strong deviation from fence counts only in 1994 for O'Ne-ell Creek. On 5 August, strip counts generated a population estimate for live sockeye (5095) that exceeded the adjusted fence count (3583) by about 42% (Table 2). This departure between methods occurred when populations were low. When few fish are on the spawning grounds, strip-count estimates are sensitive to (a) small errors in visual counts made at the strips, and (b) any bias occurring in the distribution of habitat types sampled by the strip-count stations. In 1994, strip-count stations appear to have been slightly biased toward habitats such as glides that contained above-average densities of spawners (for a discussion of errors, see Tschaplinski and Hyatt 1990, 1991). This bias appears to be consistent for O'Ne-ell Creek in 1994 and 1995, because populations estimated from strip counts were high relative to adjusted fence counts in every instance (Table 2). However, the effect of this bias was minor in 1995

Table 2. Comparison of population estimates for live sockeye spawners in O'Ne-ell Creek in 1994 and 1995 by strip counts, total-stream surveys by interval counts, and DFO fence counts. For interval counts, live sockeye were summed by 30-m intervals in 1995 and 120-m intervals in 1994. Fence counts were the cumulative number of live sockeye that had migrated through the fence up to the survey date adjusted by adding the number of live sockeye counted downstream of the fence on the same survey date, then subtracting the cumulative number of sockeye carcasses enumerated seasonally up to that same time. Adjustments were based upon DFO stream surveys conducted on alternate days.

		Population estimate method			
Year	Stream	Survey date	Strip-count expanson	Total-stream survey by interval counts	Adjusted fence counts
1994	O'Ne-ell Creek	5 August	5 095	3 068	3 583
1995 Early season	O'Ne-ell Creek	4 August	27 755	21 206	23 857
1995 Near peak of spawning	O'Ne-ell Creek	11 August	20 402	12 875	18 063

when sockeye returns were high: the population estimates obtained from the two methods were similar.

The degree of correspondence usually observed in this study between population estimates generated from fence and strip counts is relatively unusual, and indicates that strip-count estimates are relatively accurate. The accuracy of the strip-count technique increases when (a) conditions for visual counts of spawners are good, and (b) the full range of habitats throughout the spawning grounds are well represented by the strip-count stations (Tschaplinski and Hyatt 1990, 1991). These conditions were satisfied within the present study. Conditions for visual counts (relatively shallow and clear water plus bright daylight) were almost always ideal. Additionally, population surveys were based upon large numbers of stations: as many as 81, 115, and 161 stations were used to sample Gluskie, Forfar, and O'Ne-ell creeks, respectively.

The strip-count procedure also generated sockeye abundance estimates that were on average at least as accurate or better than those derived from labor-intensive, total-stream surveys (Table 2). In 1994 and 1995, total-stream surveys (interval counts) underestimated the true population size in every case (Table 2). Furthermore, the total-count procedure seriously underestimated the abundance of sockeye relative to adjusted fence counts by about 40% near the peak of spawning in 1995 (Table 2; 11 August). Total-area surveys of spawners often underestimate the true size of the population because of the difficulty in obtaining accurate counts when spawners occur in dense aggregations frequently encounterzed in pools and glides, especially in years of high fish abundance (Tschaplinski and Hyatt 1990, 1991).

Strip counts and complete (interval) counts generated population abundance estimates that differed substantially in magnitude (Table 2). However, the spatial distribution of spawners described by each technique corresponded closely. The longitudinal range inhabited by adult sockeye, and the mode and tails of the population distribution determined from strip-count samples were reflected by the same parameters determined from the spatially-continuous interval counts (compare Fig. 3 with 4, 5 with 6, and 7 with 8 respectively for O'Ne-ell Creek). This correspondence was maintained regardless of whether populations levels were low as in 1994 (Figs. 3, 4) or several fold greater as in 1995 (Figs. 5 to 8). [Visual comparisons between techniques are more difficult for 1994 because complete counts in that year were made over 120-m intervals instead of the 30-m interval generally employed in this study.]

Interval counts were superior to strip counts for illustrating the distribution of tagged sockeye (Figs. 3 to 8). The numbers of tagged fish were relatively small, especially when spawner returns were high (e.g., ca. 500 yellow-tagged sockeye in O'Ne-ell Creek among nearly 24 000 untagged fish early in 1995). Counts made at 1-m wide strips spaced 30-m apart did not efficiently sample tagged fish sparsely dispersed over several kilometres of stream. By contrast,

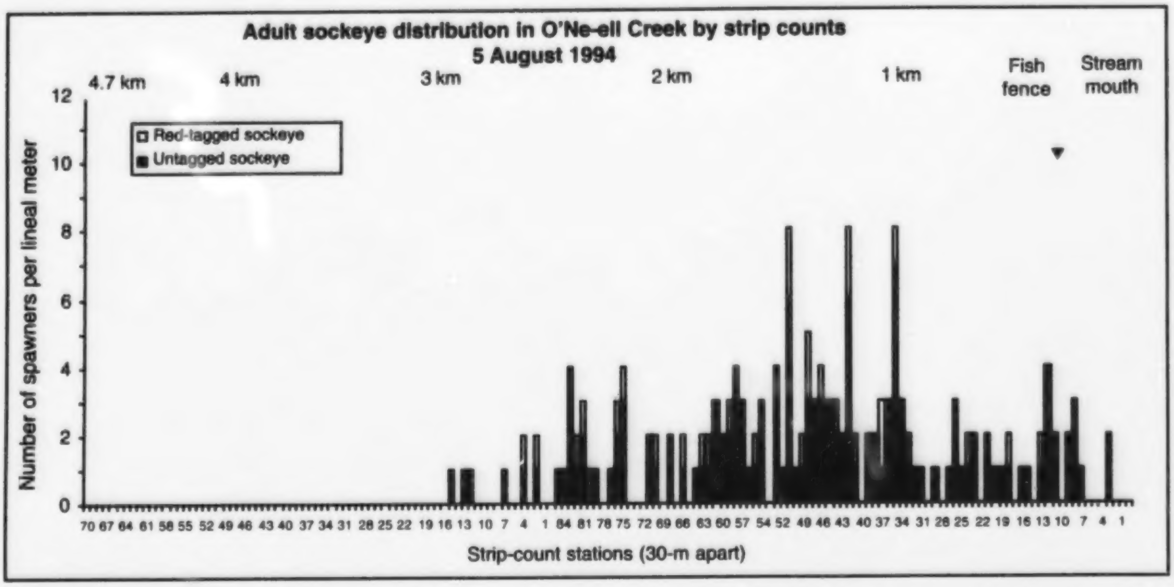

Figure 3. Longitudinal distribution of adult sockeye in O'Ne-ell Creek in 1994 by the strip-count method near the peak of spawning activity. Counts of live sockeye at each station also represent densities of fish per lineal meter. Strip-count stations were numbered consecutively within two study sections. Section 1 was located from strip 0 near the stream mouth to strip 85 (>2.6 km upstream). Section 2, located immediately upstream from section 1, had strips consecutively numbered 0–70. The historical limit of sockeye spawners was about 4.7 km upstream measured from strip 0, section 1.

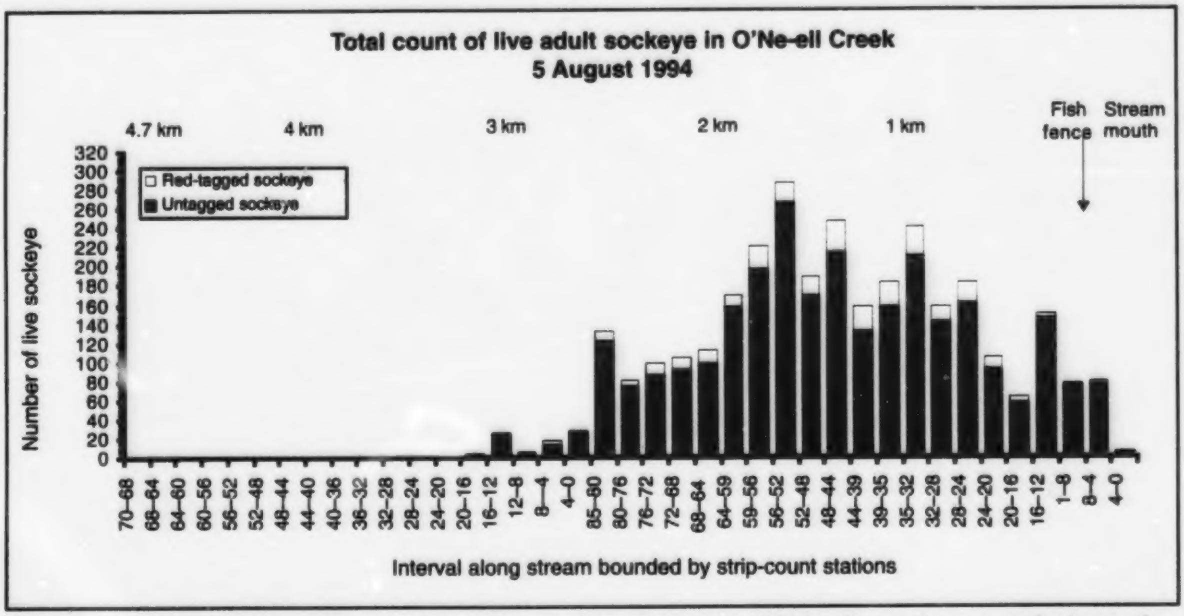

Figure 4. Longitudinal distribution of adult sockeye in O'Ne-ell Creek in 1994 by the total-count method near the peak of spawning activity. In 1994, counts of fish were summed usually by 120-m interval along the stream (a 30-m interval was used in other years).

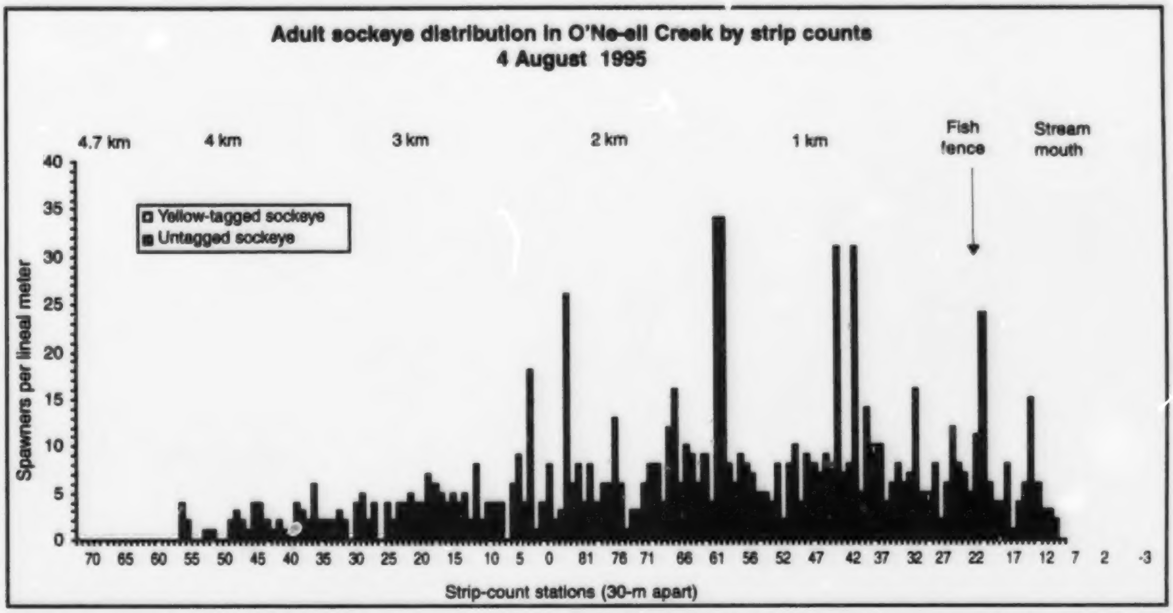

Figure 5. **Longitudinal distribution of adult sockeye in O'Ne-eil Creek by the strip-count method early in the spawning season in 1995.**

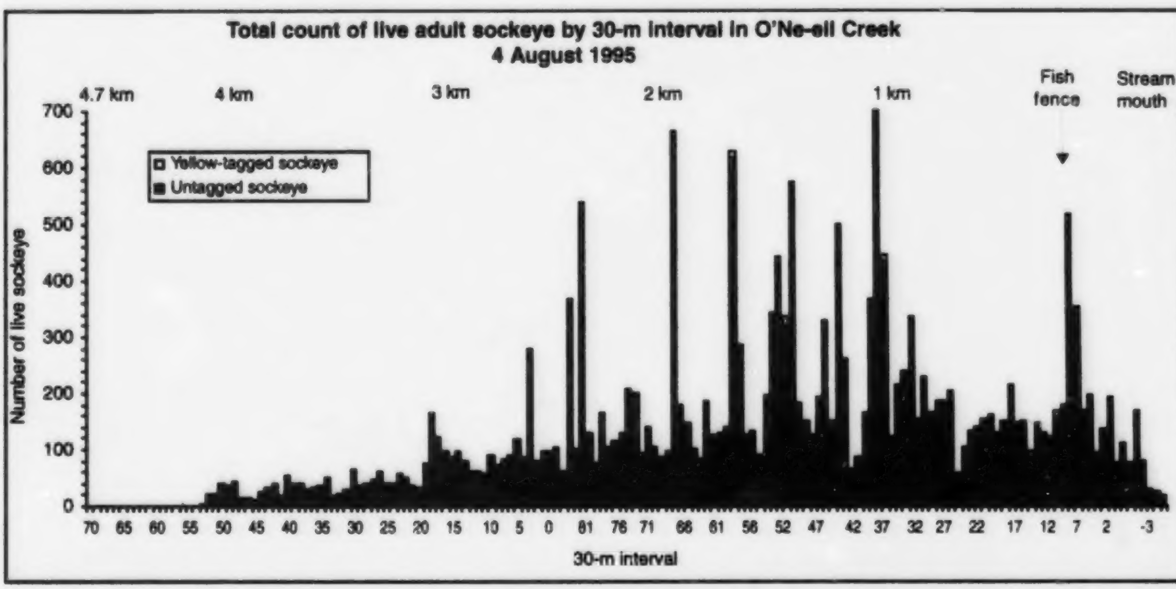

Figure 6. **Longitudinal distribution of adult sockeye in O'Ne-eil Creek by the total-count method early in the spawning season in 1995.** Counts of fish were summed by 30-m interval along the stream.

Table 3. Sockeye density (number of live fish per lineal metre) correlated with microhabitat variables for selected strips when population abundance was low. Linear correlation coefficients (r) are provided. No significant correlations were observed between spawner density and any measured microhabitat variable in 1994 (all $p > 0.05$). All strip-count stations selected were located in the lowermost reaches of each stream (section 1) where nearly all spawners resided when mean densities were low. The number of strips sampled in each stream is shown (n).

			Linear correlation coefficients (r) (spawner density vs. physical variables)			
Stream	Year	Population abundance on survey date	Stream width (m)	Water depth (m)	Strip volume (m^3)	Velocity (m/s)
Gluskie Creek (n = 11)	1994 (4 August)	3 197	–0.05	0.24	0.30	–0.40
Forfar Creek (n = 9)	1994 (5 August)	4 462	0.54	–0.21	0.11	–0.27
O'Ne-ell Creek (n = 11)	1994 (5 August)	3 583	0.17	0.07	0.55	–0.03

a majority of tagged individuals were enumerated by the spatially continuous interval-count procedure. However, the accuracy of interval counts was also limited. Even fish marked with brightly colored tags were difficult to see when tags were obscured within dense groups of untagged spawners. Counting efficiency was so reduced by this effect that only about 61–65% of tagged, live sockeye could be detected in the stream 3 days after tags were applied early in the spawning season (e.g., Figs. 4, 6). Because these percentages were determined early in the spawning season, mortality of tagged fish was insignificant. Reductions in counting efficiency can seriously reduce the accuracy of any abundance estimates based upon recovery of tagged spawners (Tschaplinski and Hyatt 1991). [Abundance estimates based upon counts of tagged sockeye are not presented here.]

In spite of some limitations, the overall performance of the strip-count method was good. Because strip-count surveys can be completed in less than one-half the time required for complete counts (Tschaplinski and Hyatt 1990, 1991), strip counts were clearly an appropriate and convenient technique to sample and study the distribution of sockeye spawners and stream habitats.

Macrohabitat-Level Assessment

Two basic patterns of spawner distribution have become clear at the macrohabitat or stream-reach level, and each pattern is associated with a different level of spawner abundance. In 1992 and 1994 when relatively low numbers of sockeye returned to the spawning grounds (Table 1), the longitudinal distribution of spawners was limited primarily to the lowermost portion of each stream (Figs. 3, 4). Seasonal estimates of spawner abundance for Gluskie, Forfar, and O'Ne-ell creeks in 1994 were only 3919, 4902, and 4371, respectively (DFO fence counts adjusted by stream surveys of spawners downstream of the fences). Mean spawner densities in 1994 were < 3 fish/lineal m in any stream, and <2 fish/lineal m in O'Ne-ell Creek (Table 1).

Because of these low numbers, sockeye occupied only part of their potential range in each creek that year. For example, spawners in O'Ne-ell Creek occupied approximately the lowest 3 km of stream (Figs. 3, 4) where gradients varied sequentially from 0.7 to 2.0%, and sediment textures changed from gravel with sand to primarily gravel with cobbles, and finally to cobbles with gravel. Within this lowermost stream area, channel morphology was controlled by LWD, featured gravel bars, and consisted of repeating sequences of riffles, pools, and glides. These characteristics of the lowermost part of O'Ne-ell Creek were repeated in the other two study streams (*see* Hogan et al. 1998).

Sockeye distribution in Gluskie and Forfar creeks in 1994 was similarly limited to the lowermost part of each stream, where the basic reach characteristics described for O'Ne-ell Creek were repeated. The same general patterns of sockeye distribution had

Table 4. Sockeye density (number of live fish per lineal metre) correlated with microhabitat variables for selected strips when population abundance was high. Linear correlation coefficients (r) are provided. Each significant correlation ($p < 0.05$) is shown with an asterisk, and provided with the r-square value (proportion of observed variance explained). All strip-count stations selected were located in the lowermost reaches of each stream where gradients were 2% or less and most (about 80–93%) spawners resided even in years when mean densities were high. The number of strips sampled in each stream is shown (n).

			Linear correlation coefficients (r) (spawner density vs. physical variables)			
Stream	Year	Population abundance on survey date	Stream width (m)	Water depth (m)	Strip volume (m^3)	Velocity (m/s)
Gluskie Creek (n = 11)	1993 (9 August)	19 556	0.78 * (r^2: 0.61)	-0.18	0.66 * (r^2: 0.44)	-0.59
Forfar Creek (n = 10)	1993 (10 August)	15 990	0.76 * (r^2: 0.58)	-0.07	0.49	-0.48
O'Ne-ell Creek (n = 11)	1993 (11 August)	26 225	0.52	0.43	0.55	0.60 * (r^2: 0.36)
Gluskie Creek (n = 14)	1995 (10 August)	9 510	0.21	0.31	0.51	-0.45
Forfar Creek (n = 11)	1995 (12 August)	10 804	0.69 * (r^2: 0.48)	-0.02	0.51	-0.57
O'Ne-ell Creek (n = 11)	1995 (11 August)	18 063	0.72 * (r^2: 0.52)	-0.42	-0.19	-0.18

been observed in 1992 when seasonal population abundances estimated by DFO were 3749, 12 904, and 11 002 for Gluskie, Forfar, and O'Ne-ell creeks, respectively (Table 1). Populations in both Forfar and O'Ne-ell creeks were more than twice as abundant in that year when compared with 1994; however, sufficient spawning habitat was apparently available in the lowermost area of each stream because only small numbers of sockeye moved upstream to spawn in the steeper-gradient sites (Tschaplinski 1994). [Spawner densities shown for Forfar and O'Ne-ell creeks in 1992 were only about 40 and 60% of their respective seasonal maxima because the strip-count surveys were conducted about 3 days after the peak of live spawner abundance. However, stream surveys by DFO staff confirmed the limited distribution of fish in both streams.]

In contrast with 1994, adult sockeye returns to all streams were markedly higher in 1993 and 1995 (Table 1). Populations in both Gluskie and O'Ne-ell creeks also increased by about two- to three-fold in 1993 and 1995 relative to 1992. Compared with returns in 1992, modest increases were also observed in sockeye abundance in Forfar Creek (about 24% higher in 1993, and 15% in 1995). In 1993 and 1995, mean densities of sockeye increased in the lowermost study section of each stream to between 4.4 and 10.0 fish/lineal m depending upon stream and year.

With these increases in density, sockeye spawners in all streams extended their ranges upstream into narrower, faster-flowing, higher-gradient reaches (>2–6%) where channel morphology was controlled more by boulders, large cobbles, and bedrock than by LWD. Substrates suitable for spawning frequently occurred as patches of gravel and cobbles located among the boulders. For example, sockeye mean density in the lowermost part of O'Ne-ell Creek increased from 1.8 fish/lineal m in 1994 (Fig. 3) to 6.1 fish/lineal m in 1995 (Fig. 5). Accordingly, spawners extended their range by >1 km to occupy about 4.3 km of the stream measured from strip-zero located 240 m downstream of the fish fence (Figs. 5, 6). In 1993, when peak numbers of live fish exceeded 26 000, sockeye occupied their maximum range in O'Ne-ell Creek which extends about 4.7 km upstream from strip 0 (see strip 70, section 2; Fig. 5).

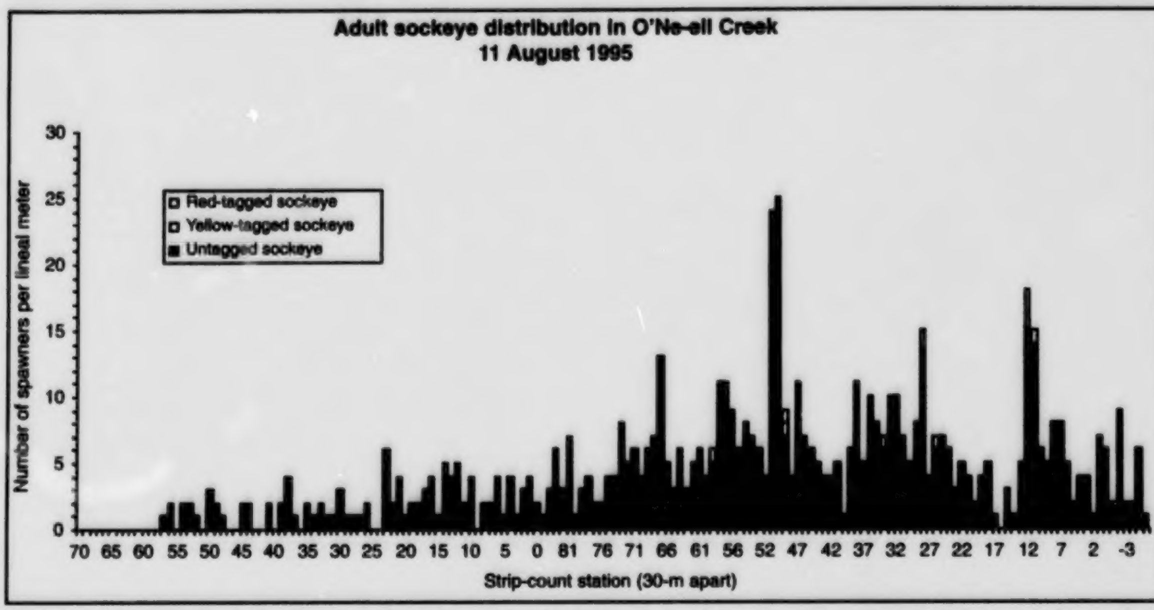

Figure 7. Longitudinal distribution of adult sockeye in O'Ne-ell Creek by the strip-count method near the peak of spawning activity in 1995 (roughly in mid-season).

Physical barriers to migration limit the distribution of sockeye spawners in all streams. In Forfar Creek, this barrier is a waterfall located only about 750 m upstream from the lowermost study section, which contains almost all sockeye in years of relatively low spawner abundance. In both O'Ne-ell and Gluskie creeks, the barriers are located within canyons containing stepped-pool, boulder cascades. Sockeye in both streams usually are limited to the lower parts of the canyon where gradients are <8%. In Gluskie Creek, the upstream limit of sockeye distribution is only about 2.5 km upstream from the stream mouth.

Mesohabitat-Level Assessment

Analyses of sockeye habitat preferences at the mesohabitat level are yet in progress. Therefore, the results of the categorical data analyses, including logistic regressions of spawner density versus the habitat type surveyed at each strip are not currently available. However, some general trends are apparent. Unusually high sockeye counts were frequently obtained from major pools where large aggregations of sockeye occurred. These schools were especially common early in the spawning season and consisted primarily of individuals that recently arrived on the spawning grounds (as shown by the abundance of recently tagged fish), which are not yet fully mature. Previously, these sockeye were in holding pools prior to moving out from them to spawn.

Holding pools were important habitats both in years of low and high spawner abundance. For example, in 1994 when relatively few sockeye returned, the highest spawner densities occurred in pools at strips 35 and 51 at the peak of spawning in O'Ne-ell Creek (8 fish/lineal m; Fig. 3). In 1995 when sockeye returns were over five-fold higher, peaks associated with pools occurred at strips 28, 31, and 50 early in the spawning season (section 1 downstream, 24–34 fish/lineal m; Fig. 5). Peak counts also occurred later near the height of spawning activity at some of the same strips in 1995 (e.g., 28, 50, and 51; Fig. 7).

Many of the strips located in these pools also contained complex habitats, which included laminar-flowing glides, especially if the strips were situated at the downstream margins of the pools (e.g., O'Ne-ell Creek, strips 50 and 51; Figs. 5, 7). Although some sockeye spawned within the pools, the highest densities of fish that were actually spawning were most often found at strips that contained these pool-and-glide elements, or that were located entirely within gravel glides. Most of the secondary peaks in abundance (as well as some of the primary ones) occurred at strips located in glide habitats. Examples of these glide-associated peaks are strips 12 and 42 in 1994 (Fig. 3; section 1), and strips 11, 12, 57, 58, 66 in 1995 (Fig. 5; section 1).

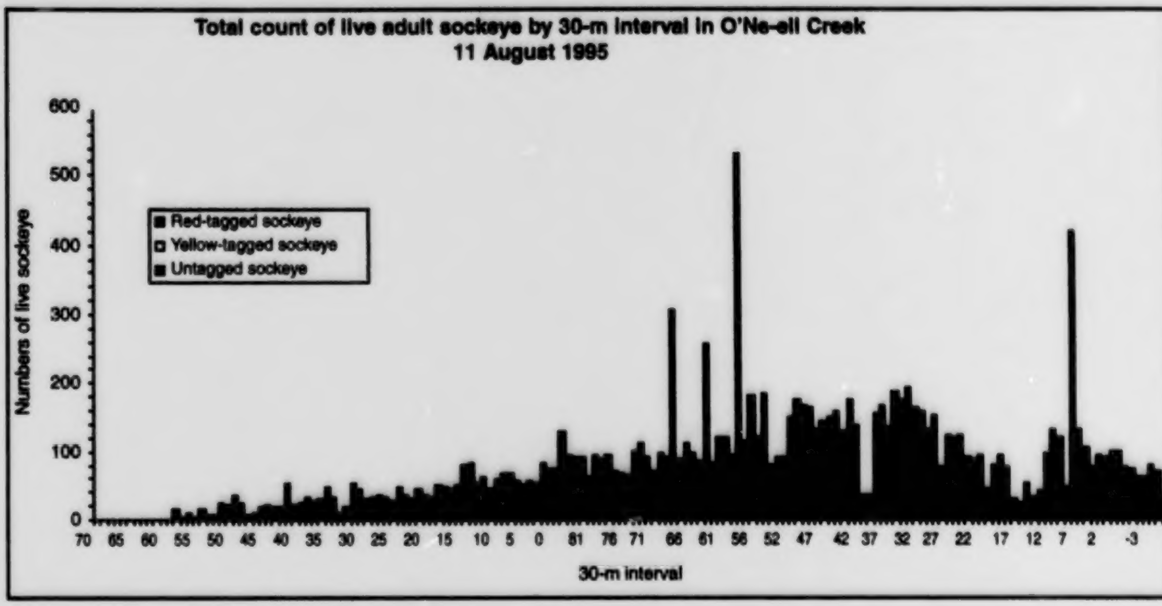

Figure 8. Longitudinal distribution of adult sockeye in O'Ne-ell Creek by the total-count method near the peak of spawning activity in 1995 (roughly in mid-season). Counts of fish were summed by 30-m interval along the stream.

Despite these general observations, variations occurred both within and between years. For example, several glide sites contained only average numbers of spawners in any year. Additionally, the relative importance of some sites changed radically at different times during the season. Densities at strip 25 (section 1) in O'Ne-ell Creek fell from 26 fish/lineal m on 4 August (Fig. 5) to only 4 fish/lineal m 1 week later near the peak of spawning activity (Fig. 7). High densities of spawners also occurred in sites other than glides and the tails of pools. Sockeye were sometimes abundant in gravel-and-cobble riffles (e.g., strip 75, Figs. 3, 5). Additional observations and continued analyses of sockeye densities and habitat categories are required to clarify spatial and temporal variations in spawner distribution.

Although some strips were associated with unusually high numbers of sockeye, spawner densities at most strips were near average for each stream (Figs. 3, 5, 7; Table 1; Tschaplinski 1994). Despite local peaks in abundance, adult sockeye were relatively evenly distributed throughout the spawning grounds. The even distribution of sockeye within each stream reach partly explains the relatively high accuracy of spawner abundance estimates generated from strip counts which sample the population from evenly spaced enumeration stations (Tschaplinski and Hyatt 1990, 1991). More significantly, the observed distribution of adult sockeye also suggests that high-quality spawning habitat was available virtually everywhere, at least in the lowermost reaches of each stream. When habitats in the lower parts of each stream contained the maximum densities of fish they could support, spawners moved into areas upstream (Figs. 5, 6, 7, 8). The only sites that sockeye appeared to avoid were areas flooded by beavers (e.g., strips 15–16, section 1; Figs. 7, 8) and steep, fast-flowing, riffles and rapids in sites upstream (section 2, strips 46–47, 50–51; Figs. 7, 8).

Microhabitat-Level Assessment

In 1994, when sockeye abundance was low, spawner densities were not correlated significantly with any measured microhabitat variable including water depth, velocity, or indices of habitat space such as stream width and strip volume (Table 3; all $p > 0.05$). These observations confirm similar results obtained in 1992 when spawner abundance was again relatively low (Tschaplinski 1994). When relatively few adult sockeye return to the study streams, mean densities of fish are low, and spawner distribution is not limited by available space. Sockeye are free to select prime spawning sites from less favorable ones. However, the observed range of spawner densities was also low in the same years. This relatively even distribution of fish indicates that habitat

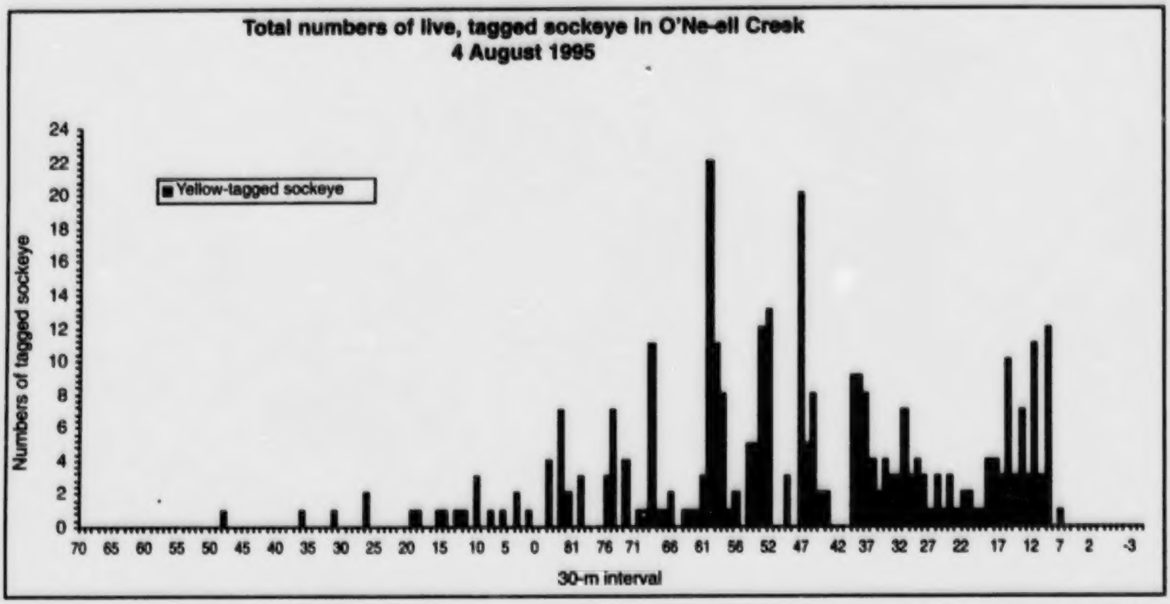

Figure 9. Longitudinal distribution of yellow-tagged sockeye 3 days after tags were applied to migrants at the fish fence early in the spawning season. Counts of tagged fish were summed for each 30-m interval along the stream. The sample of tagged fish was well mixed within the spawning population. Tagged sockeye quickly dispersed throughout the stream to reflect the range and distribution of untagged sockeye.

space in the lowermost stream reaches was plentiful, active site selection was limited, and the full range of depths, velocities, and spawning substrates were acceptable to the spawners. The results of surface-substrate texture analyses are not yet available; however, analyses of gravel samples collected by the frozen-core method confirms the high quality of spawning and egg-incubation substrates throughout the lower reaches of the study streams (Scrivener 1994). Furthermore, studies have shown that egg-to-fry survival is exceptionally high throughout the lowermost reaches of all study streams, and is the equivalent of survivals obtained in the best artificial spawning channels (Cope and Macdonald 1998).

In years when spawner abundance is high, some statistically significant correlations between sockeye densities and spatial indices were observed (Table 4). For example, spawner densities and stream width were significantly and positively correlated in Gluskie and Forfar creeks in 1993, and in Forfar and O'Ne-ell creeks in 1995 (Table 4; all $p < 0.05$). Spawner distribution (density) was thus a function of available habitat space when population levels were high. These results confirm observations of spawner distribution at the mesohabitat and macrohabitat levels; that is, when sockeye are abundant, available habitats in the lowermost portion of each stream are fully occupied, and spawners then expand their distribution upstream into steeper-gradient areas.

One significant, positive correlation has been observed between spawner density and habitat volume (Gluskie Creek, 1993; $p < 0.05$), and spawner density and velocity (O'Ne-ell Creek, 1993; $p < 0.05$; Table 4). Because of variation among years and streams, interpretation of these isolated results is difficult. However, the significant correlation between velocity and sockeye densities in O'Ne-ell Creek in 1993 likely resulted from spawners avoiding extensive stretches in the lowermost part of that stream where velocities were low because of beaver dams.

Distribution of Tagged Sockeye and Competition for Space

Observations on spawner distribution and habitat use, the high quality of the streambed substrates, and high egg-to-fry survival rates within the Stuart–Takla study (Cope and Macdonald 1998) clearly illustrate that the three principal study streams are exceptionally productive for sockeye salmon. However, results of tagging studies suggest that spawners can limit their own production within these streams.

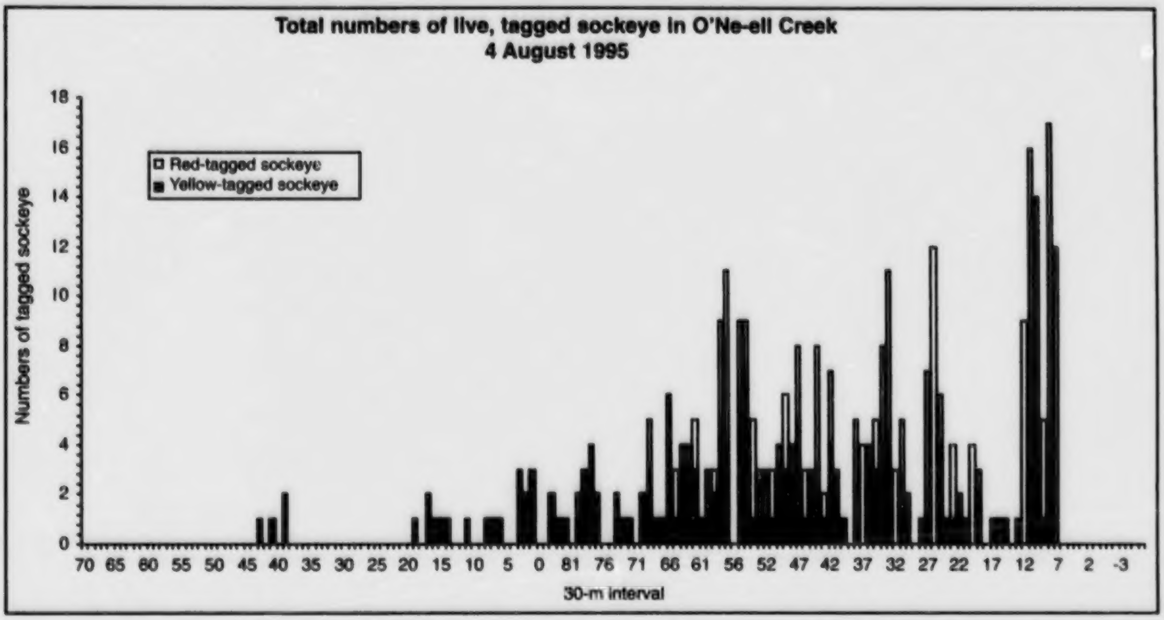

Figure 10. Longitudinal distribution of yellow-tagged and red-tagged sockeye 3 days after red tags were applied to migrants at the fish fence close to the peak of spawning activity. Counts of tagged fish were summed for each 30-m interval along the stream. The sample of yellow-tagged fish was still well mixed within the spawning population; however, all red-tagged fish remained in the lower portion of the stream (section 1).

Sockeye tagged early in the spawning season in 1995 quickly migrated and dispersed upstream as far as the untagged individuals. Three days after tagging, the distribution of tagged fish was equal to that of the untagged ones (Fig. 9). One week later, the distribution of the same tagged fish had not changed (Fig. 10; yellow-tagged sockeye). They were less abundant due to mortality among those that had already spawned. However, the fish that remained alive were actively spawning on top of redds belonging to fish that had spawned first. Superimposition of redds was confirmed through observation of large numbers of dead salmon eggs excavated by later spawners. Dead eggs drifting downstream with the current are especially common in years of high spawner abundance.

Furthermore, sockeye that arrived 1 week after the first sample of fish was tagged also spawned on top of redds belonging to the earlier arrivals (Fig. 10; red-tagged sockeye in O'Ne-ell Creek). Tagged samples of fish arriving in mid-season (or later) did not migrate as far upstream as the earliest arrivals. The distribution of fish arriving in mid-season was almost entirely towards the lowermost stream reaches (Fig. 10). In years of high spawner abundance, these low-gradient portions of the stream usually contain 84–94% of all spawners. Competition for space by superimposition of redds appears to be common in sites downstream although more area of high-quality habitat is available in that part of the stream compared to those reaches found upstream (e.g., section 2 upstream; Figs. 9, 10).

Sites upstream are occupied primarily by sockeye arriving prior to mid-season. Although suitable spawning substrate in the higher-gradient reaches is limited and frequently occurs in patches among boulders, egg-to-fry survivals in these areas are as high as those observed in sites downstream (Cope and Macdonald 1998). The upper reaches of the spawning grounds support at most only about 6–16% of all sockeye. However, fry production from these sites might be disproportionately large compared to the proportion of adults spawning in the same areas assuming redd superposition causes high mortality in the principal spawning sites downstream.

Summary

The results of studies on adult sockeye distribution and habitat preferences prior to forest harvesting have shown that:

1) The strip-count method can be employed to determine the abundance and longitudinal distribution of adult sockeye salmon in Gluskie, Forfar, and O'Ne-ell (Kynoch) creeks.
2) Large numbers of sockeye aggregate in deep pools in all streams prior to spawning.
3) The highest densities of spawners during the peak of spawning activity are found most frequently in gravel glides and at the downstream tails of pools.
4) Of the three study streams, the only areas sockeye consistently avoid are sites flooded by beavers, and high-velocity riffles and rapids found in the steep-gradient reaches upstream.
5) The length of stream occupied by adult sockeye depends directly upon overall levels of spawner abundance.
6) Linear correlations between spawner densities and stream microhabitat variables indicate that the entire range of depths, velocities, and (by extension) substrate textures occurring in the lowest stream reaches are suitable for sockeye spawners.
7) Substrate analyses and high egg-to-fry survival in all sites sampled by other observers confirm the abundance of high-quality spawning sites throughout the lower reaches of each stream accessible to sockeye.
8) Sockeye arriving later in the spawning season do not migrate as far upstream as the earliest arrivals: these fish may limit the survival of eggs deposited by early spawners by superimposition of redds.

Population and habitat assessments will continue in the future to determine (a) during-harvest and postharvest changes in habitat type, stream channel structure and stability, and the spatial distribution of adult sockeye, and (b) the consequences of these changes upon sockeye egg-to-fry survival and fry production from the study streams.

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Temperatures in Aquatic Habitats: the Impacts of Forest Harvesting and the Biological Consequences to Sockeye Salmon Incubation Habitats in the Interior of B.C.

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Abstract

The impacts on stream temperature as a result of forest harvesting in riparian areas have been documented in a number of locations in North America. The effects of these impacts on the development rates and emergent behavior of juvenile salmonids is not as well understood. As part of the Takla Fishery/Forestry Interaction Study, we have investigated the influence of forest harvesting on stream temperatures in northern British Columbia. We have documented the influence of stream temperatures on embryo development rates and the timing of fry emigration from natal streams. These preliminary results show that temperature also has an indirect effect on fry emigration characteristics through its influence on the timing and duration of snow-melt discharge events in the spring.

Introduction

The early run of Stuart River sockeye salmon (*Oncorhynchus nerka*) spawn in the Takla tributaries during early August when annual stream temperatures are at their warmest and water levels are at their lowest summer levels (Macdonald et al. 1992). They complete a 1150-km migration up the Fraser River system to reach their spawning grounds (Burgner 1991). The natural variability of stream temperature regimes and the influence of temperature on spawning and incubation success must be understood before any impacts from forest harvesting can be identified and quantified. Baseline stream temperature data have been collected continuously from one harvested (Bivouac Creek) and three unperturbed (Gluskie, Forfar, and Kynoch creeks) watersheds in the Takla Lake region since 1990 (Scrivener and Andersen 1994a). During many years prior to 1990, the Pacific Salmon Commission and Department of Fisheries and Oceans have recorded summer stream temperatures from these and other Takla tributaries.

Macdonald, J.S., Scrivener, J.C., Patterson, D.A., and Dixon-Warren, A. 1998. Temperatures in aquatic habitats: the impacts of forest harvesting and the biological consequences to sockeye salmon incubation habitats in the interior of B.C. Pages 313–324 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

From previous studies conducted in coastal and interior settings, we can predict that summer stream temperatures will rise in association with streamside clearcut logging (Brown and Krygier 1970; Brownlee et al. 1988). The effects on winter stream temperature regimes have been documented (Hartman and Scrivener 1990) but are poorly understood, particularly in northern interior watersheds. The protective influence of riparian management zones as prescribed in the Forest Practices Code of British Columbia (FPC of B.C.), is as yet unproven, particularly in small tributary systems where little streamside vegetation remains after forest harvesting (S4, S6 streams; British Columbia Ministry of Forests and British Columbia Ministry of Environment 1995).

A number of authors have examined the biological consequences of elevated stream temperatures for the spawning, fertilization, development, and emergence of salmon eggs and fry. Exposure to elevated temperatures during spawning migrations can increase metabolic rates and stress levels, and deplete energy reserves, which may a) increase prespawning mortality (Gilhousen 1990); b) create temperature barriers to migration (Gilhousen 1990; Williams 1992); c) reduce gonad development (Bouck et al. 1975); and d) reduce fertilization success due to reduced sperm mobility (Smirnov 1976) or egg viability (Campbell 1992, 1994). High temperatures can also increase the susceptibility of salmonids to disease (Groberg et al. 1978) and decrease the interval between infection and death (Sanders et al. 1978).

Growth and development rates and the eventual emergence timing of salmon embryos are stock specific and are influenced by the temperature of the incubation environment (Godin 1981; Murray and McPhail 1988). Spawning time has evolved so that embryos will emerge from the gravel in coincidence with spring food production in their rearing habitat (Bams 1969; Brannon 1987). Northern sockeye salmon stocks, such as those in the Takla Lake region, spawn earlier in the year than southern stocks, have a greater rate of embryonic development, and hatch at a smaller size (Beacham and Murray 1989). Mechanisms such as these compensate for the wide variety of temperatures including long periods near freezing that occur naturally during the incubation period and ensure that time of emergence is similar to stocks that incubate in warmer conditions (Brannon 1987; Scrivener and Andersen 1994a).

In the Takla Lake region, forest harvesting is likely to have the greatest impact on stream temperatures in the late summer during early embryonic development. However, natural conditions in northern interior streams during the mid-winter are poorly studied, which makes it difficult to anticipate potential impacts of harvesting. It is unclear whether embryos from stocks that have evolved to compensate for wide temperature ranges can compensate for elevated temperatures associated with forest harvesting. Increased development rates and higher metabolic activity may cause premature exhaustion of their energy reserves; untimely emergence from the gravel; and early emigration from natal streams (Scrivener and Andersen 1994a). Increases in temperature related to harvesting during summer, fall, and winter in Carnation Creek on the west coast of Vancouver Island were a causal factor in the reduction of incubation and rearing periods and caused earlier emigration of coho salmon (*O. kisutch*) and chum salmon (*O. keta*) (Holtby and Scrivener 1989; Hartman and Scrivener 1990).

This paper provides a summary of preliminary investigations of stream temperature impacts in harvested watersheds in the northern interior of B.C. with reference to unharvested watersheds in the Takla Lake area. Of particular importance is the documentation of impacts on stream temperatures during cooler months of the year when the loss of streamside vegetation allows increased radiation loss to the atmosphere. We address potential biological consequences of stream-temperature impacts on sockeye salmon by presenting preliminary information indicating that embryo development rates and timing of fry emigration from natal streams are dependent on stream temperatures during incubation. These preliminary results show that temperature also has an indirect effect on fry emigration characteristics through its influence on the timing and duration of snow-melt discharge events in the spring.

Methods

The Takla Fishery/Forestry study design and study location was described by Macdonald et al. (1992), Macdonald (1994), and Macdonald and Herunter (1998). Stream temperature measurements from the Takla experimental watersheds provide a description of baseline inter-annual temperature variability as these watersheds are largely unharvested. Temperatures were collected between 1990 and 1995, from three of the experimental tributaries

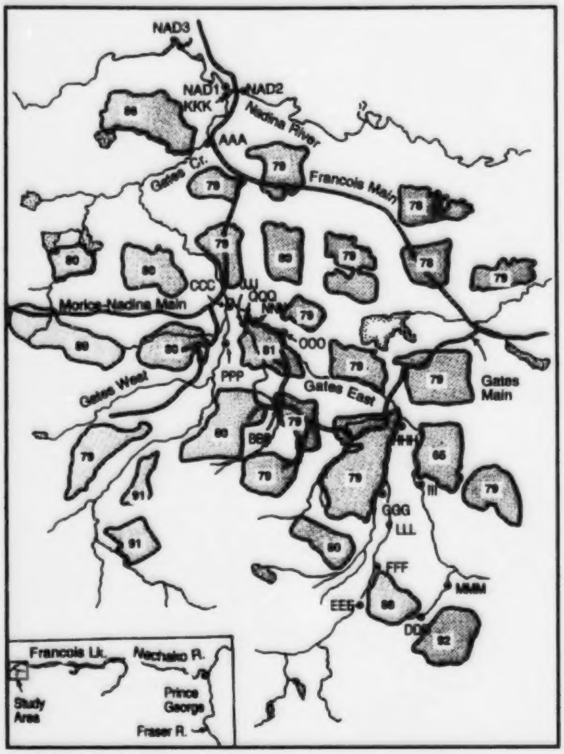

Figure 1. Cutblock locations and harvesting dates in the Gates Creek watershed. Temperature recording began in the summer of 1993, and continued during the summers and falls of 1994 and 1995 at locations located with a solid circle. Records were collected at many of the locations through the winter of 1995. A weather station (i.e., air and water temperatures, solar radiation) was located at site BBB.

(Gluskie, Forfar, and Kynoch creeks) [map provided in Macdonald and Herunter (1998)]. Gates Creek, a tributary to the Nadina River (about 200 km south of the Takla Lake watersheds), was used to measure the impacts of past forest harvesting (Fig. 1). Harvesting started in the Gates Creek watershed in 1966, and extensive forestry activities have continued to the present time. Stream temperature measurements have been collected during the summer and fall from 1993 to 1996. The Gates Creek study design and location is described by Andersen et al. (1997).

Stream temperature measurements were obtained from electronic data logging equipment. Originally, Unidata model 6003 were installed on the streambanks. They were placed in insulated containers and buried in the ground in the winter to ensure that batteries continued to function in the extreme cold. During the last 2 years, the Unidata equipment was replaced with self-contained Vemco data loggers, which operate while completely immersed in the stream. In the Takla experimental watersheds, thermistors were placed in the stream at numerous locations between the stream mouths and the main data acquisition site 1–2 km upstream (above the road bridge on each stream) (Macdonald et al. 1992). After spawning activities in late summer, thermistors were placed in several redds in each stream at depths of 10–20 cm below the surface of the gravel to document inter-gravel temperatures through the incubation period. A 5-year daily stream temperature average from three creeks was plotted in combination with data from a single creek during a single year. A summation of the mean daily temperatures (°CTU) from three creeks over 4 years during the incubation period was calculated.

To examine the temperature impacts associated with forest harvesting, thermistors were placed in selected tributaries of Gates Creek above and below cut blocks with known harvest and silvicultural histories. Sites were chosen to ensure that data were collected from a range of stream sizes (classified according to the FPC of B.C.) with a variety of aspects and topographies (Fig. 1). This paper will consider stream temperatures in the east and west forks of Gates Creek (site CCC) and in a tributary to East Gates Creek that flows through a 1988 cut block (sites EEE, FFF, LLL). At site CCC, Gates Creek is about 15 m wide (bankfull width) and, using FPC of B.C. guidelines, is classified as an S2 stream. The west fork watershed has been harvested the least and the stream has had more riparian protection than the east fork (Table 1). At sites EEE, FFF, and LLL, Gates Creek has an S6 classification (<3 m width, without fish), has a channel that is not deeply

Table 1. A 1995 comparison of watershed size and harvesting activities in the west and east forks of Gates Creek. Most harvesting began in the late 1970s and continues today.

	Gates west	Gates east
Watershed area (km^2)	37	24
Percent cut	10	26
Length of stream exposed (km)	1.9	8.6
Average leave strip width (m)	53	47
Average age of cutblock (yr)	13	15

incised, and since harvesting has a 500-m exposure, which faces an easterly direction. In the Takla Lake streams and in Gates Creek, temperatures were collected from all sites at 0.5- or 1.0-hour intervals and were entered into a database as daily means, maximums, and minimums (Andersen et al. 1997). Weather stations have collected air temperature and solar radiation information at Takla Lake since 1990, and at Gates Creek since 1993 (Fig. 1, station BBB). Data exploration is restricted to visual interpretation/comparison of graphical displays. Time-series analysis and temperature-response modeling will occur in the future.

A laboratory experiment was performed between August 1994 and May 1995 to examine embryo development rates of sockeye salmon at two temperature regimes. A control regime mimicked temperature patterns observed in the Takla experimental tributaries (Scrivener and Andersen 1994a). A second temperature regime mimicked forest harvesting temperature impacts by the addition of 1.8°C to the water temperature during the early incubation period (August 4–November 1). Temperature impact predictions were based on observations from Gates Creek.

Eggs and sperm were collected from mature sockeye salmon at the mouth of Gluskie Creek and then flown to the West Vancouver Laboratory, West Vancouver, B.C., where they were fertilized and water hardened within a 12-hour period. Thirty fertilized eggs were immediately placed in a capsule (Cope and Macdonald 1998) and immersed in gravel in one of four tanks (60 L) in each temperature regime. A maximum of ten capsules were placed in each tank. Refrigerated water was recirculated through the systems and metabolites were removed by constantly replenishing a portion of the water supply. Temperatures were maintained by thermostatic control and recorded with an electronic data logger. Capsules were removed at monthly intervals and a preliminary development stage estimate was recorded. Embryos were then preserved in Stockard's solution, stored, and labeled in preparation for detailed development stage analysis (Velsen 1987). This document describes the preliminary development stage data.

Since 1990, inclined plane traps have been installed on the Gluskie, Forfar, and Kynoch creeks in the spring, to enumerate sockeye salmon fry emigrating from natal habitats to rear in local lakes (Hickey and Smith 1990). In conjunction with fecundity estimates taken during the enumeration of adults at fences at the mouths of each river, the fry data provide egg-to-fry survival and emigration timing estimates (Smith 1994). Plots of the time at which 50% of the fry emigrated from each creek, each spring versus thermal units accumulated during the corresponding incubation period have allowed us to explore the influence of natural stream temperature variability on emigration timing. In the absence of harvesting-induced temperature impacts in the Takla experimental watersheds, these plots document the effects of natural disturbance regimes and are the first step toward estimating the possible effect of forest harvesting on fry emigration.

Information pertaining to annual variation in water output from the experimental streams has been collected since 1990 (Scrivener and Andersen 1994b). These data must be considered in conjunction with stream temperatures when documenting correlates of emigration timing and duration because a strong relationship exists between increases in stream output during spring snow-melt and the emigration timing of fry (Macdonald et al. 1992). The degree to which hydrographic characteristics (i.e., timing, duration, skewness, kurtosis) influence the emigration behavior of sockeye salmon fry is examined with correlation analysis and graphical presentations.

Results

The annual pattern of stream temperatures in the Takla experimental streams (Fig. 2) is typical of records collected from other northern interior watersheds including the Nechako River (Blachut and Faulkner 1988) and tributaries to Slim Creek (Brownlee et al. 1988; Choromanski et al. 1993). Summer stream temperatures in the Takla area are between 8 and 16°C (daily means) with diurnal ranges of 1–4°C. Annual temperature maximums occur during August when peak sockeye salmon spawning occurs (Tschaplinski 1994). Temperatures remain warm through August and a portion of September, but decline rapidly to below 1°C by late October. Stream temperatures usually remain below 1°C from the third week of October to the second week of April and show little diurnal variability. Temperatures below 0°C were recorded 20 cm below the surface of the stream bed during the coldest periods in the winter (January–March) and are likely an indication of anchor-ice formation within the locations where eggs were deposited (Fig. 2). During the period of spring snow-melt, water temperatures rise slowly to approximately 5°C by late May, when most of the sockeye salmon fry have emigrated from the streams (Macdonald et al. 1992). Diurnal ranges are small (1–2°C) during the winter and spring.

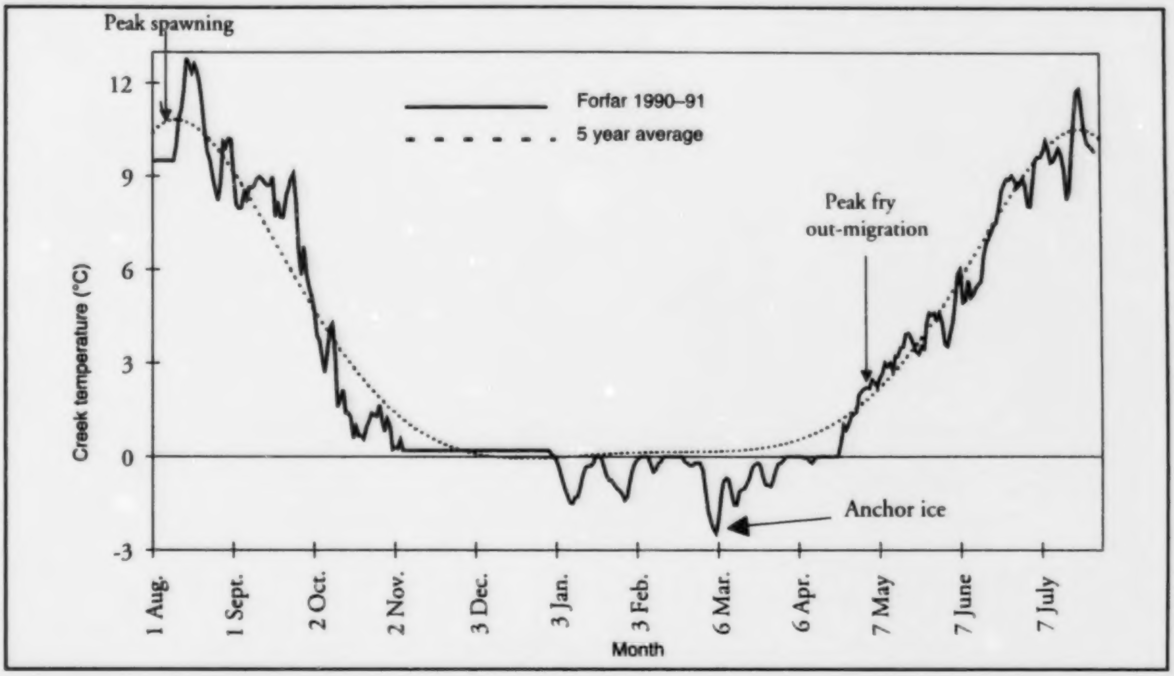

Figure 2. Annual inter-gravel thermograph data for Forfar Creek with 5 year averages, presented as a typical representation of all streams in the Takla experimental area (Scrivener and Andersen 1994a). Average sockeye peak spawning and peak fry out-migration times are indicated. Temperatures below 0°C in mid-winter, are indications of anchor-ice formation within the incubation gravel.

Temperatures in Gates Creek and associated tributaries also approached 16°C in mid-summer (Fig. 3 and 4) with the warmest temperatures being recorded in the lower reaches of Gates Creek at sites AAA and KKK (Andersen et al. 1997). Water temperatures tended to increase with decreasing elevation but summer temperatures remained several degrees cooler at the mouth of Gates Creek than in the Nadina River.

Harvesting impacts on stream temperatures were clearly demonstrated during the summer months. Stream temperatures in the more heavily harvested East Gates Creek (Table 1) were frequently higher than in West Gates Creek by as much as 2.5°C (Fig. 3). In a smaller system in the upper reaches of the Gates Creek watershed (stations EEE, FFF, LLL, Fig. 1), where a 1988 harvest created a 500-m opening adjacent to the stream, water temperatures frequently rose 3°C as a result of canopy removal (Fig. 4). Rapid reduction and partial recovery of water temperature were noted as the stream reentered forested lands (station LLL) despite no obvious input of cool water to the stream immediately below the clearcut. Diurnal temperature ranges were always greater in the portions of the watershed influenced by harvesting (Andersen et al. 1997). As air temperatures declined in the fall, the influence of riparian harvesting had the opposite effect to the influence experienced in the summer; water temperatures were cooler in the more affected system (e.g., East Gates Creek, Fig. 3).

During embryo development, sockeye salmon undergo three distinct development stages (i.e., egg, alevin, and fry). During the early incubation period (autumn, Fig. 5), 90% of the thermal unit accumulation occurs, resulting in rapid embryo development, and in some cases, hatching to the alevin stage. Little development is likely to occur after November 1, when average daily temperatures fall below 0.5°C (winter, Figs. 2 and 5). Sockeye salmon fry are in the midst of emigration from the streams (early May) before stream temperatures rise in the spring. From 1991 to 1995 in the three experimental creeks, an average of 660°CTU accumulated during the period between fertilization and fry emergence. An increase in the average daily temperature by 1.5°C between

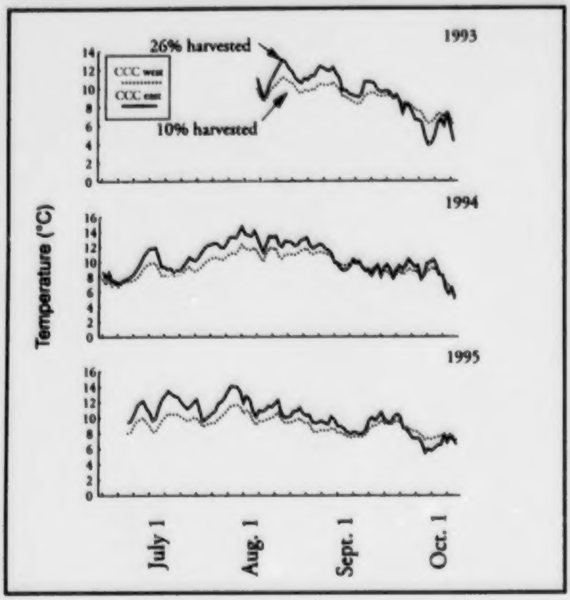

Figure 3. Summer and fall stream temperatures during 1993, 1994, and 1995 in the east and west forks of Gates Creek at monitoring station CCC (Fig. 1). Temperatures are daily averages.

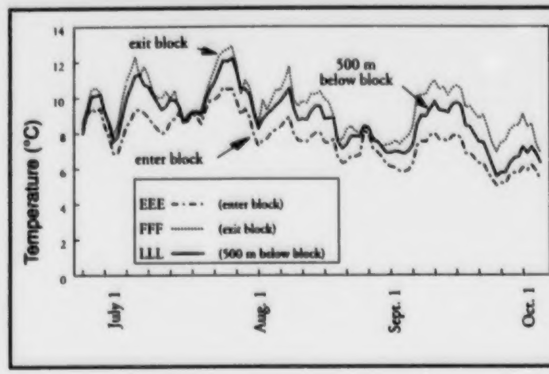

Figure 4. Daily average stream temperatures during the summer and fall of 1995 in a small tributary of the east fork of Gates Creek (Fig. 1). Temperatures were recorded as the stream entered a cutblock (E), 500 m further downstream as it re-entered the forest (F) and a further 500 m downstream within the forest. The cutblock was harvested in 1988.

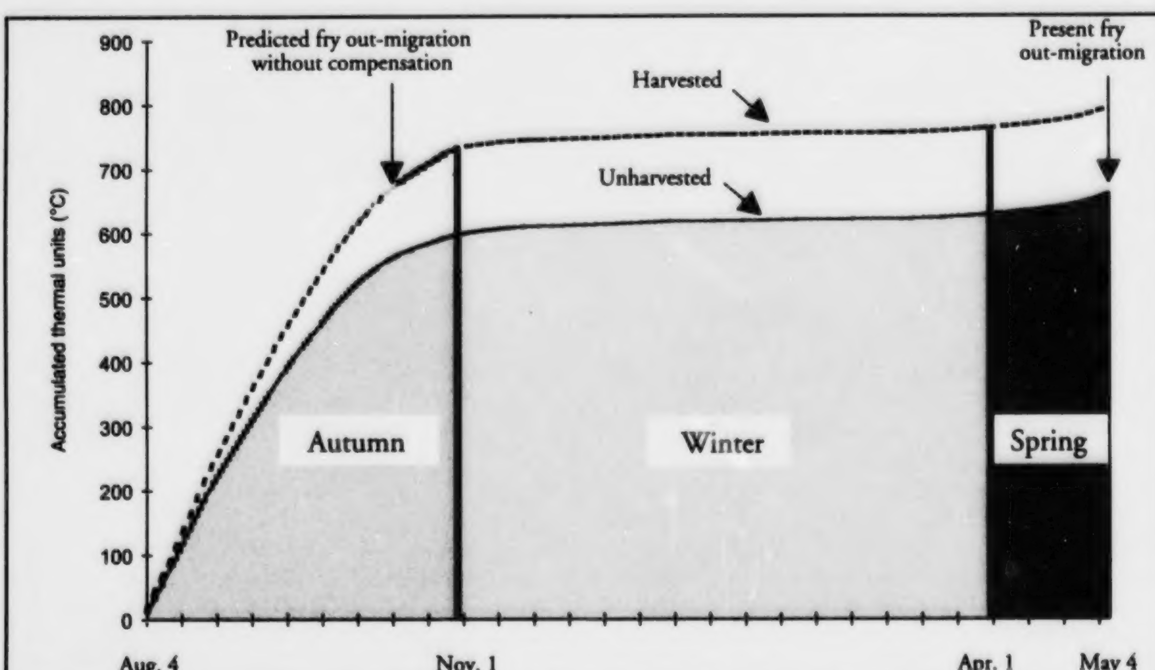

Figure 5. Accumulated mean daily temperature units (%CTU) observed in Forfar Creek (unharvested) (Fig. 2) and predicted as a result of harvesting. Harvesting is assumed to cause a 1.5°C increase in temperature each day during the autumn period (as observed in other locations in B.C.'s interior region, e.g., Gates Creek, Slim Creek) (Brownlee et al. 1988).

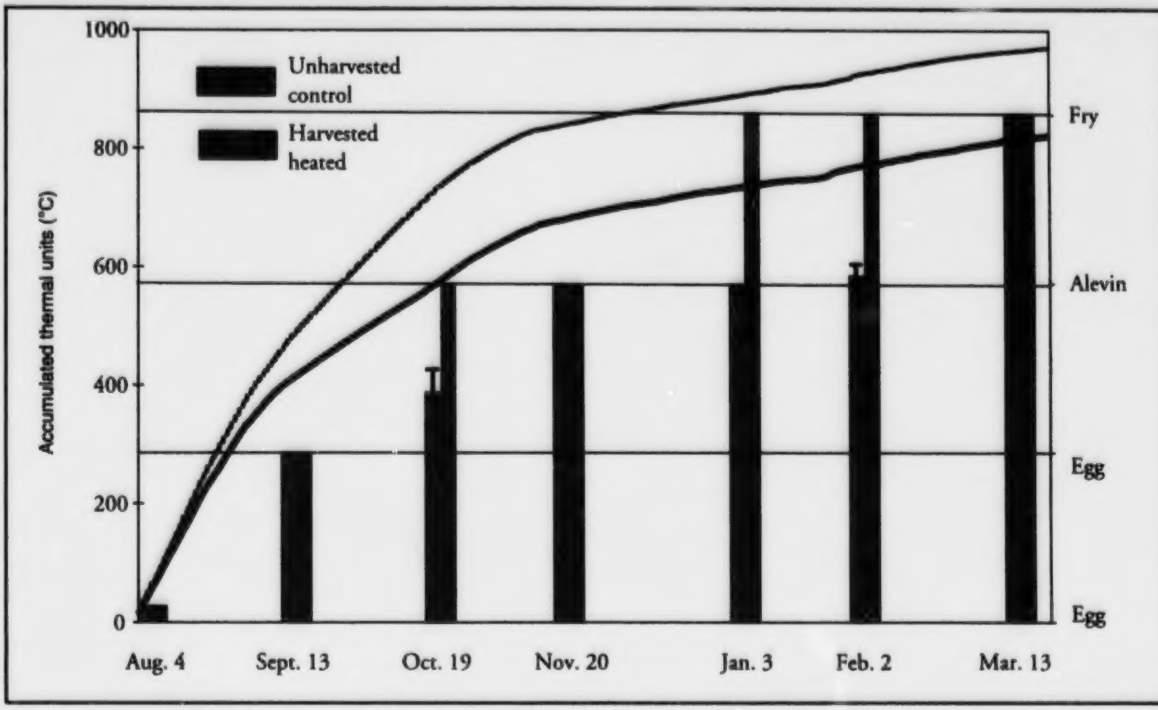

Figure 6. A laboratory analysis of egg-to-fry development at natural and harvested (heated) temperature regimes. The harvested temperature regime approximates the accumulation of thermal units as predicted from a temperature model presented in Figure 5. Error bars describe 2 SE of the mean.

August 4 and November 1 as a predicted response to riparian harvesting would result in the accumulation of about 700°CTU by late October and 794°CTU by early May (Fig. 5).

A decrease in the development time of sockeye salmon embryos will occur when temperatures in the incubation environment rise as predicted in Figure 5 and as documented in the harvested portions of Gates Creek. In the laboratory, all eggs raised under the warmer temperature regime had hatched by October 19 after receiving 737°CTU, whereas 64% of the eggs in the control regime were unhatched with 578°CTU (Fig. 6). Similarly, at the warmer temperatures, all embryos reached the fry stage by January 3 and were apparently ready to emerge. It was March 13 before 100% of the embryos from the control temperature regime reached the fry stage. It is important to note that, regardless of which temperature regime is considered, hatching to the motile alevin stage occurred by mid-November before the coldest period of the winter.

We predict that accelerated development to the fry stage will result in earlier emergence and earlier emigration from natal streams. Within the annual range of natural temperature fluctuations, fry emigration occurs earlier when average daily water temperatures are warmer (t-test, $p < 0.05$, Fig. 7). Temperatures conditions during the early incubation period (August 4–November 1, Fig. 5) have nearly the same predictive power as conditions during the entire incubation period. If forest harvesting results in warmer late summer and autumn temperatures in the Takla experimental streams, emergence and emigration may be accelerated and occur in the following spring earlier than they occur in neighboring unaffected watersheds.

A great deal of variation exists among the annual spring hydrographic patterns exhibited in the three experimental streams between 1991 and 1995 (Fig. 8). For example, in 1992, the snow-melt-generated water discharge was more protracted than it was in 1995, and was characterized by a repetitive series of discharge events that corresponded to the arrival of warm fronts and precipitation in the upper watersheds. The majority of fry emigrated from their natal streams on the rising limbs of each of the discharge events over a 1-month period. If the

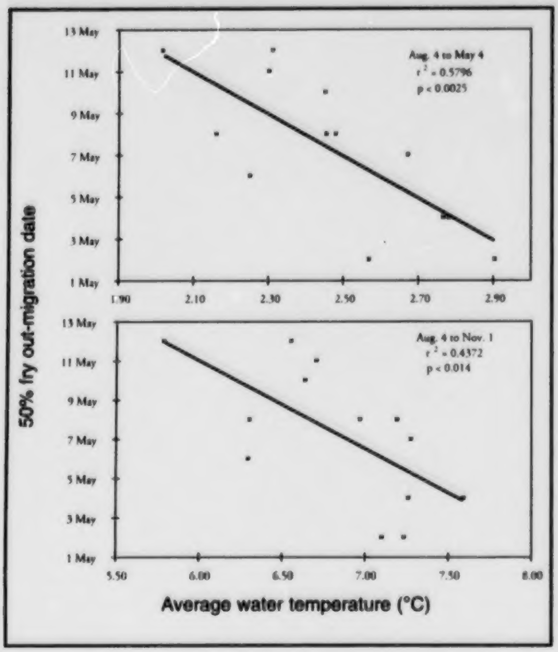

Figure 7. The effect of average daily water temperatures on the date of peak sockeye fry emigration from each of three Takla experimental creeks during a 5-year period. Daily temperatures were compiled during the entire incubation period (August–May) and during the period when most of the thermal units were accumulated (August–November).

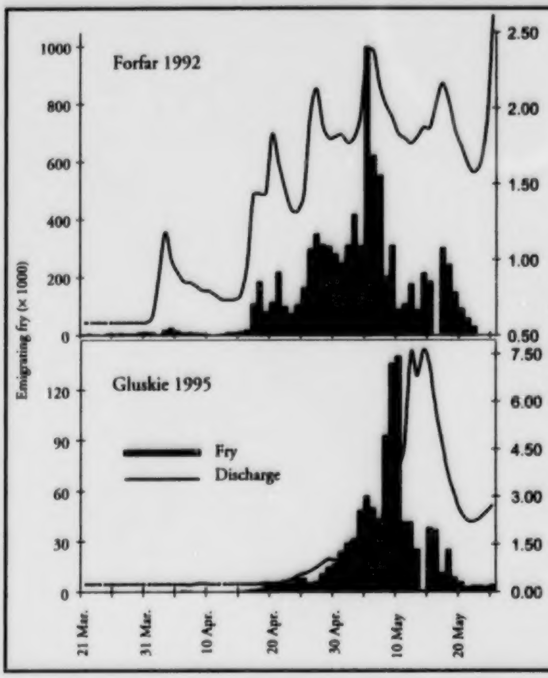

Figure 8. Two water discharge events during the spring snow-melt period, that represent the range in hydrographic conditions experienced by the sockeye fry during their emigration from the experimental streams. Fry leave on the rising limb(s) of the hydrograph.

spring water discharge patterns are more stable (as in Gluskie 1995), fry will emigrate during a narrower time frame (15–20 days) in response to a single or a few hydrographic peaks. As a result, both the variability and the skewness of the discharge curves have significant influence on the variability and skewness of the fry emigration pattern (t-test, $p<0.01$, Fig. 9).

Discussion

A plethora of information from many geographical areas emphasizes the value of riparian vegetation for the moderation of summer stream temperatures (reviewed in Anderson 1973). Stream exposure to the direct effects of solar radiation is the dominant mechanism involved in the increased temperatures, while air temperature and energy loss from the stream due to evaporation play minor roles (Brown 1970; Brown and Krygier 1970). Stream temperature reductions upon re-entry to forested areas have also been documented and are generally attributed to the inflow and mixing of cool tributary streams and groundwater (Hall and Lantz 1969; Swift and Baker 1973). Despite the absence of any obvious source of cooling water between sites EEE and LLL in the Gates Creek watershed, a moderation of stream temperature was detected. This serves to emphasize the importance of understanding of groundwater reserves and their sources when managing multiple landbase issues. Temperature impacts due to forest harvesting may be insignificant if groundwater temperatures are unaffected and if stream temperature increases remain below lethal or sub-lethal limits to the resident biota (as is the case in most portions of Gates Creek). An increase in temperatures in small tributary streams may have little impact on the cooling of mainstem, fish-bearing habitat if the total amount of water affected is relatively small in comparison to the amount from all sources in the watershed. This subject will receive additional attention

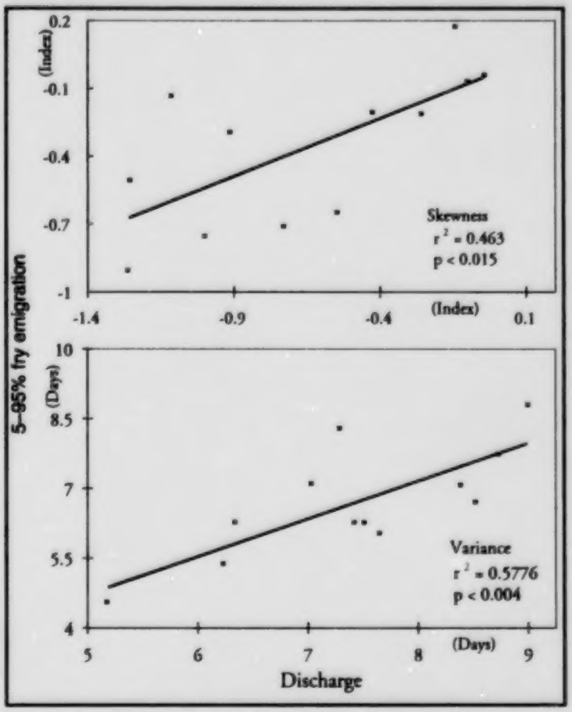

Figure 9. The effects of hydrographic variance and skewness on the variance and skewness of sockeye fry emigration distributions during the period of sockeye fry outmigration. Data have been compiled from three Takla experimental creeks during a 5-year period.

during future research activities planned for the Takla Project.

Forest harvesting impacts on stream temperatures during winter are poorly understood, particularly in northern interior locations. Meehan et al. (1969) reports little change to winter stream temperatures due to harvesting in a coastal Alaskan stream. In Carnation Creek, a coastal stream in southern B.C., winter stream temperatures were generally warmer (<1.0°C) after harvesting; these warmer temperatures are thought to have positive consequences for the development and subsequent rearing of juvenile coho salmon (Hartman and Scrivener 1990). At Carnation Creek, winter temperatures are less likely to be influenced directly by solar radiation (due to shorter days and reduced solar angles) and are more likely influenced by changes to groundwater temperatures and levels (Shepherd et al. 1986; Hartman and Scrivener 1990). In northern interior environments, cold winter air temperatures in conjunction with the loss of the insulating qualities associated with riparian vegetation may promote radiant and/or conductive loss of energy from the stream to the atmosphere at a greater rate in harvested than unharvested systems (Anderson 1973). This seasonal reversal in the effect of forestry activities may have consequences unique to northern aquatic biota such as reduced production periods, reduced water levels and habitat area, earlier ice cover, and possible increased anchor-ice formation.

The direct relationship between incubation temperature and embryonic development is well documented (Brannon 1987; Murray and McPhail 1988). Early spawning Stuart River sockeye salmon, the stock that spawn in the Takla experimental streams in early August, require fewer thermal units to develop to the fry stage than most other stocks of salmon (660 vs. >800°CTU, Velsen 1987; Murray and McPhail 1988). At warmer temperatures the relative rate of development will decline to compensate for both colder climates and the effects of natural interannual variation in climatic conditions experienced during the incubation period (Brannon 1987). This provides a mechanism to offset the effects of reduced availability of temperature units in northern climates (Scrivener and Andersen 1994a). Daily mean temperatures varied approximately 2°C (August to November) among the creeks and years examined in this study, but peak fry emigration dates were no more than 2 weeks apart. However, this compensation mechanism may not be able to adjust for the impacts of forest harvesting during warm years if temperature impacts follow patterns documented at Slim (Brownlee et al. 1988) and Gates creeks (e.g., increases of 1.5–2.0°C). Accelerated rates of development may result in developed fry being available as early as January and in excess of 700°CTU being accumulated by late October. At warmer temperatures, yolk reserves may be insufficient to meet their metabolic needs (Scrivener and Andersen 1994a). Fry mortality may result from as little as 2 weeks of starvation (Bilton and Robins 1973). If emergence from the gravel were to occur in mid-winter, survival in their natal stream or in their rearing habitat would be doubtful. Fry emigration has evolved to coincide with spring zooplankton production in their rearing habitat (Bams 1969; Shepard 1984).

These temperature and development data must be interpreted with caution. Natural conditions during the mid-winter period, in northern interior streams are poorly understood. Calculations of mean daily temperatures were made from stream temperatures and not from the temperatures within the

gravel incubation environments. Stream temperatures may not accurately represent the thermal regime experienced by developing embryos due to factors associated with groundwater intrusion and surface water infiltration (Shepherd et al. 1986; Macdonald 1994). Of equal importance is the mobility of embryos in late October during the alevin stage. Mobility during the onset of winter permits the alevins to seek out conditions more conducive to successful development. Bams (1969) documented alevin movement within incubation environments in response to reduced water flow and temperature. Dill (1969) describes downward movement in the gravel following hatching in coho salmon and rainbow trout (*O. mykiss*), which may be a reaction to avoid light or to seek warmer water temperatures, which increase with increasing depth in the gravel (Macdonald, unpublished data). However, mobility entails an energetic cost, which is manifested in reduced size at emergence (Dill 1969).

Temperature has an indirect influence on fry emigration timing in the spring as a correlate of snowmelt and stream discharge. A number of authors have reported multimodal emigration patterns in synchrony with peaks in stream discharge (reviewed by Godin 1981). Downstream migrations by sockeye salmon may be active (McCart 1967) or passive (Bams 1969), frequently occur at night (Godin 1981), and may be in response to current velocity (Arnold 1974). The role of current velocity may be the dislodging of the fry from the gravel or other stream features that provide cover (Godin 1981), or increasing current velocities may act as a cue to synchronize emigration; these questions merit further study. A rapid increase in water discharge and high water levels have been shown to create synchronous hatching in coregonids (Naesje et al. 1995), while a varying flood pattern gave protracted hatching periods. Future research in the Takla experimental watersheds will address the role of discharge in controlling emigration timing.

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Water Temperature Dynamics in Small Forested Headwater Streams of Newfoundland, Canada: Quantification of Thermal Brook Trout Habitat to Address Initial Effects of Forest Harvesting

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Abstract

Studies in small headwater streams of the Copper Lake Watershed, Newfoundland, Canada, have described temperature characteristics and defined temporal dynamics of fluvial habitats for brook trout (*Salvelinus fontinalis*). Paired Hugrun thermographs have recorded hourly water temperatures at the upper and lower end of each of five stream reaches from 1993 to 1995. Four of these reaches are in headwater streams and the fifth is a second order stream. These sites are scheduled for different buffer zone treatments as part of an experimental study on the effects of forest harvesting. Temperature dynamics through the watershed in 1995 demonstrated a general cooling trend in the early summer months (June and July) that shifted to a warming trend, likely related to the warming and stratification of standing waters in the drainage basin, in August and September. Monthly mean and maximum summer water temperatures indicated that unharvested fluvial habitats provided a cooling benefit with decreased daily means, maxima, and diel variations over the reaches. Limited harvesting on one stream resulted in increased daily mean and maximum temperatures and increased diel variations as contrasted with unperturbed streams. Hourly summer temperatures were quantified in the context of temperature preferenda and stress and lethal limits for brook trout and were used to calculate a summer thermal habitat suitability index (HSI). The HSIs indicated the fluvial habitats were thermally suitable for brook trout, varying from 0.72 to 0.82. The change in temperature dynamics in the harvested reach did not result in a significant change in thermal HSI (decrease from 0.82 to 0.80 over the reach).

Scruton, D.A., Clarke, K.D., and Cole, L.J. 1998. Water temperature dynamics in small forested headwater streams of Newfoundland, Canada: quantification of thermal brook trout habitat to address initial effects of forest harvesting. Pages 325-336 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

In Newfoundland, intensive forest harvesting activities have been ongoing since the early 1900s. Much of the commercial timber in Newfoundland is associated with riparian areas; consequently, potential interaction between fish resources and forestry practices is high (Newfoundland Forest Service 1994). The effects of forest harvesting, if any, on wild fish populations in the region are not well understood. Harvesting activity frequently occurs in watersheds containing small headwater streams, which are important nursery areas for resident salmonids, particularly brook trout (*Salvelinus fontinalis*) (Power 1980). These headwater areas, which typically consist of streams with finer substrates and relatively stable hydrological regimes, are important for salmonid production and successful spawning and rearing (Murray and Harmon 1969).

A number of important life processes for fish, including growth, reproduction, physiology and ultimately survival, are linked to water temperature (Fry 1971). Stream water temperature regimes can be altered by forest harvesting and significant increases in summer water temperatures have been reported in a number of studies (e.g., Beschta and Lynch 1988; Brown and Krygier 1970; Lynch et al. 1977, 1984; Meehan 1970). These increases in temperatures can exceed the preference ranges or physiological tolerances of resident fish (e.g., Brown and Krygier 1970; Lee and Samuel 1976; Murphy et al. 1986; Scrivner and Andersen 1984) and can influence the relative productivity of aquatic habitats. Stream temperature is related to water volume; therefore small, shallow headwater streams heat up much faster than larger streams and rivers. Consequently, harvesting activities can potentially have a larger impact on headwater systems and can lead to cumulative temperature increases in downstream reaches.

Much of the existing knowledge about fishery–forestry interactions is based on research carried out on the west coast of North America where fish fauna, climate, forest conditions, and bio-physiography are radically different from Atlantic Canada. Consequently, the results of these studies can only generally be transferred for use with boreal forest ecosystems in eastern Canada and for fish species and riverine habitats in Atlantic Canada. The current recommended forest management practices for environmental protection in Newfoundland (Scruton et al. 1996) are based on best available information and lack region-specific considerations, such as topography, soil type, hydrologic regime, water quality, climate, fish species, and habitat use.

In order to address the need for region-specific infonnation, a large multi-agency, multi-disciplinary study has been initiated. The *Copper Lake Buffer Zone Study*, undertaken within the Western Newfoundland Model Forest, is intended to help determine an effective buffer-strip width that will afford protection for fish and wildlife habitats and maintain water quality (Scruton et al. 1995). This study is intended to provide the scientific rationale and/or criteria to categorize habitat disturbances arising from forest harvesting activities and to demonstrate the benefits of mandatory riparian leave strips. This will permit the classification of habitats, considering sensitivity to impacts, which then can be used to evaluate effects on aquatic resources in order to identify reasonable trade-offs between environmental and economic sustainability of the forest industry.

Brook trout are studied due to their wide distribution throughout Newfoundland, recreational importance to residents, and relative sensitivity to the potential effects from forest harvesting (e.g., temperature, sedimentation, etc.). Stream temperature is a basic limiting factor for poikilotherm species (Brett 1956) and maximum summer water temperatures are considered the most important factor limiting the distribution of brook trout (MacCrimmon and Campbell 1969, Barton et al. 1985). Increasingly, changes in temperatures and other water quality parameters have restricted brook trout to headwater streams (MacCrimmon and Campbell 1969). A major component of this study therefore includes assessing the temperature regime of the fluvial habitats in the watershed and evaluating the effect of forest harvesting with prescribed buffer strip treatments. This paper presents the initial results from the water temperature component of this study and describes the pre-harvest characteristics of the study reaches. The approach for determining and assessing change in associated thermal habitat quality is presented. The initial effects of harvesting in one stream reach is described.

Materials and Methods

Study Area

The Copper Lake Watershed, with a drainage basin of 13.5 km^2 and elevations of 350 to 650 masl, is a small headwater system consisting of a number of small lakes and streams (Fig. 1). Stream gradients and riparian slopes range from moderate to high (Table 1). The watershed is typical of heavily forested and rugged topography areas in western

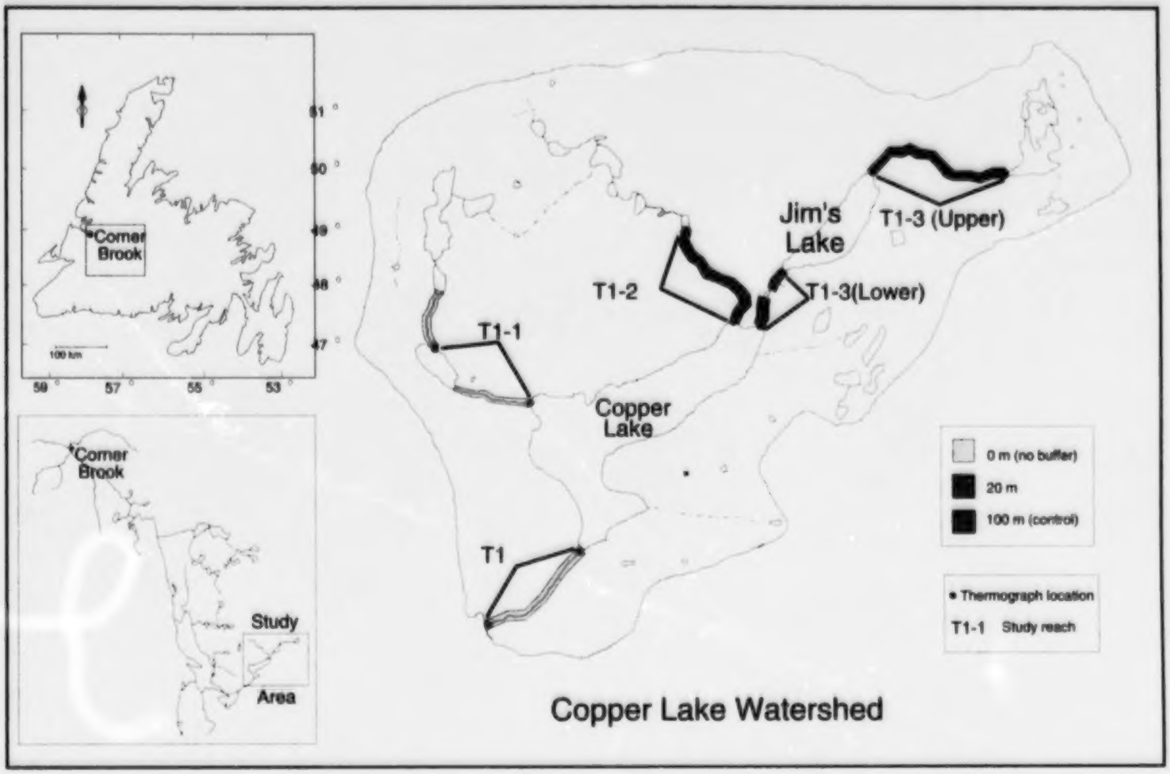

Figure 1. Location of the Copper Lake Watershed in insular Newfoundland, Canada, including the study reaches, thermograph locations, and proposed harvesting (buffer zone) treatments.

Newfoundland and has never been harvested. The forest comprises mature and overmature balsam fir (*Abies balsmea*) with some intermixed black spruce (*Picea mariana*). Surface soils are dominated by glacial tills. The susceptibility of the watershed to forest harvesting (based on soil erodibility, moisture content, and slope) is considered high to extreme (van Kesteren 1992).

Fluvial habitats within the watershed consisted of 5 headwater streams (T1-1 through T1-5) and the Copper Lake outlet (T1, a second order stream); of which T1, T1-1, T1-2, and T1-3 contained suitable brook trout habitat and were selected for detailed study. Selected physical characteristics of the study reaches, including stream order, length, mean width, mean depth, and proportion of habitat types, and canopy cover (after Gibson et al. 1987), are provided in Table 1. Standing waters in the catchment, important for temperature regulation (Bartholow 1989), included a total of 134.1 ha of lake area. Copper Lake was the largest water body at 82.4 ha. Tributary T1-1 contained four lakes totalling 15.2 ha while T1-2 contained six smaller lakes totalling only 6.5 ha. Tributary T1-3 contained a total of 30.0 ha, all of which were situated above the lower reach and 12.5 ha above the upper reach. Jim's Lake (17.5 ha) is located between the upper and lower reaches of T1-3.

Mean annual flows for the study reaches were estimated at 0.55, 0.08, 0.08, and 0.14 m^3s^{-1} for T1, T1-1, T1-2, and T1-3, respectively. Seasonal distribution of flows would be similar to other drainages in western Newfoundland with a low flow period in the summer (July to September) and winter (January to April) and peak flow periods in the spring (May and June) and late fall (October to December) (Newfoundland Department of the Environment and Lands 1992). Spring peak flows, in association with snow melt, are about twice the magnitude of fall peak flows. The climate of the region was characterized as cool (average annual temperature of 5.2°C) and moderately wet (average annual precipitation of 1186 mm). The warmest months are July and August with average (57 year) air temperatures of 17.4 and 16.8°C, respectively (Environment Canada 1991).

Table 1. Characteristics of the study reaches (between thermograph locations)

Study reach	Stream order	Length (m)	Mean width (m)	Mean depth (cm)	Gradient (m/km)	Habitat types (%)	Canopy cover (%)
Trib. T1 (Copper L. outlet)	2	1440	8.45	14.2	50.1	Riffle (66) Rapids (31) Flat (2) Steady (1) Other (1)	27.7
Trib. T1-1	1	795	3.35	6.5	36.5	Riffle (91) Steady (6) Other (3)	45
Trib. T1-2	1	850	6.27	8.5	88.2	Riffle (75) Rapids (2) Steady (1) Run (1) Other (21)	53.8
Trib. T1-3 (Lower), below Jim's Lake	1	200	6.15	7.5	25.5	Riffle (100)	87.5
Trib. T1-3 (Upper), above Jim's Lake	1	1070	3.07	8.3	66.7	Riffle (70) Rapids (3)	15.8

Study Design

The overall objective of the Copper Lake Buffer Zone Study is to determine the benefits of protection of productive capacity of fish habitat associated with maintaining an unharvested strip of timber along the riparian zones of salmonid habitats, with a primary focus on the fluvial component (Scruton et al. 1995). This involves contrasting physical, chemical and biological attributes in stream reaches subject to various harvesting treatments including: i) no buffer strip (harvesting to the stream edge), to identify maximum impacts and to provide a historical perspective on past activities, T1 and T1- 1; ii) harvesting under the currently prescribed no-harvest buffer strip (20 m plus a slope factor), lower reaches of T1-2 and T1-3; and iii) harvesting with a strip width of 100 m (control reaches), upper reaches of T1-3 (Fig. 1). No harvesting was conducted in the study catchments in the first and second years (from 1993 to the early fall of 1994) to allow collection of baseline data. Limited harvesting (1.82 ha, 20% of the watershed, 9% of the stream length) was conducted in the fall of 1994 in the lower reaches of T1-1 (no buffer). Year three (1995) was the first year of assessment after initial cutting and extensive harvesting is scheduled for 1996.

Temperature Recording

Thermographs were deployed in pairs at the upper and lower end of each of five stream reaches to evaluate the temperature change over the stream length in relation to the prescribed buffer-zone treatment. The study reaches included: i) T1, the Copper Lake outlet, and the primary (1°) tributaries ii) T1-1; iii) T1-2; iv) T1-3 Lower and v) T1-3 Upper (Fig. 1). Stream temperatures were recorded hourly using Hugrun recording thermographs (Type A, temperature range –2°C to +38°C, accuracy 0.1°C). The hourly recordings were used to calculate daily and monthly means, daily and monthly minima and maxima, and diel ranges. As we were primarily concerned with potential stream warming, we paid most attention to the period from June 1 to September 30, when temperatures typically exceed 5°C and reach annual maxima. Battery failure [T1-1 (Upper) and T1-3L (Lower) in 1994] and loss of a thermograph [T1 (lower) in 1995] resulted in missing data.

Temperature dynamics and thermo-regulation provided by the forest canopy cover for each stream reach were determined by comparing data from the upper and lower thermographs. Two reaches are examined in detail to provide an evaluation of the

initial effects of harvesting on stream temperatures including T1-1 (harvested treatment) and T1-2 (control – no harvesting). Comparisons included changes in monthly mean and maxima, daily mean and maxima, and in diel variation (Bartholow 1989).

Thermal Habitat

Because the existing and post-harvesting temperature regimes may influence the suitability of fluvial habitats, a number of strata were defined based on published temperature requirements of brook trout (e.g., Jirka and Homa 1990; McRae and Edwards 1994; Meisner 1990; Power 1980; Raleigh 1982). These strata were delineated as follows:

i) < 11°C (Lower; below optimum but not stressful);

ii) 11 to 16°C (Optimum; preferred range with good growth potential);

iii) 16 to 21°C (Upper; above optimum but not stressful);

iv) 21 to 24°C (Stress; potential stressful condition, energy reserves used for thermo-regulation, poor growth potential, increased susceptibility to other stressors such as disease); and

v) above 24°C (Lethal; potentially lethal temperatures if exposed for a period of time).

For each thermograph, the total number of hours in each of the five strata were calculated (Table 2). These data were also used to develop a single index, or summer thermal habitat suitability, using the habitat suitability index (HSI) curve for temperature in Jirka and Homa (1990). An HSI was calculated for each hourly recording and a mean HSI value was then determined for the summer period (June 1 to September 30) for each location and year (Table 2).

Results

Monthly mean and maximum temperatures for the summer months (June to September) indicate that the dominant influence of each of the reaches on stream temperature was to provide a cooling effect (Figs. 2 and 3). In 1995, temperatures were reduced at the lower end of each reach, compared with the upper station. These differences were most notable in T1-2, which had a high proportion of canopy cover (54%), and T1-3U, the longest reach (1070 m) of the primary study streams. An exception to this observation in 1995 was for T1-1, the treatment stream, where harvesting was conducted in the late fall of 1994, and monthly mean and maximum temperatures were observed to have increased over the reach.

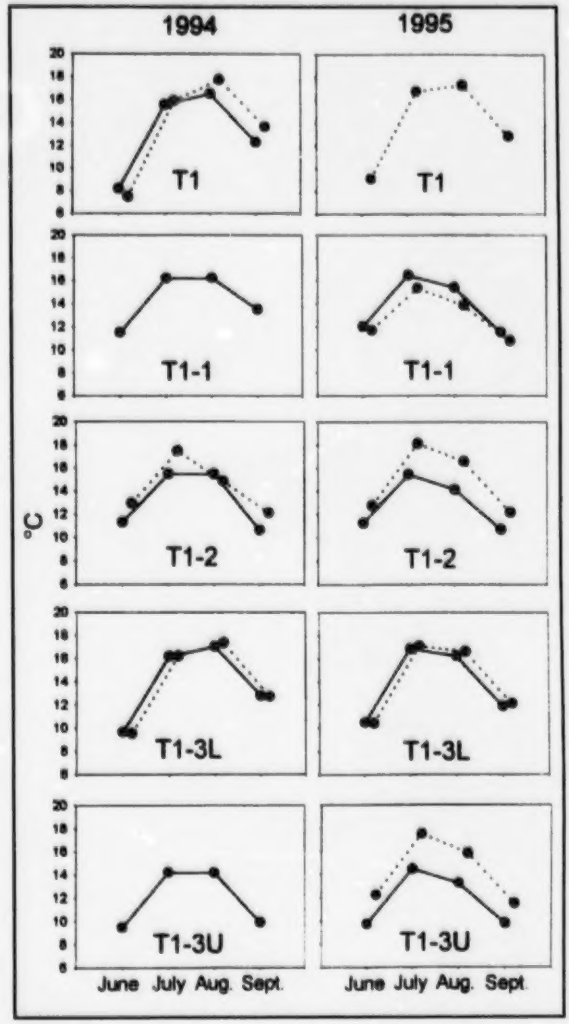

Figure 2. Mean summer monthly temperatures in the upper (dotted line) and lower (solid line) stations for each of the five study reaches in 1994 and 1995.

The temperature dynamics in the unharvested reaches of the watershed in 1995 were examined through comparison of changes in mean monthly temperatures from the upper watershed (T1-3U[B]) through to the Copper Lake outlet (T1U) (Fig. 4). There was a general cooling trend in the early summer months (June and July), which shifted to a warming trend in August and September. The cooling effect was evident throughout the summer in the unharvested fluvial reaches. Increases in temperature were evident through the major standing water

Table 2. Proportion of time summer (June 1 to September 30) water temperatures fell into five temperature ranges to define thermal brook trout habitat

Study reach	Thermo-graph	Year	Total hours	Lower (<11°C)	Optimum (11 to 16°C)	Upper (16.1 to 20.9°C)	Stress (21.0 to 23.9°C)	Lethal (>24°C)	Thermal summer habitat suitability[a]
			Total hours and percentage of time in temperature range						
T1									
	T1-(Lower)	1993	686	0 (0)	402 (59)	277(40)	4 (1)	3 (0)	0.85
		1994	2928	642 (22)	1139 (39)	834(29)	4 (0)	0 (0)	0.79
		1995	2928	-	-	-	-	-	-
	T1-(Upper)	1993	1250	66 (5)	638 (57)	484 (39)	62 (5)	0 (0)	0.80
		1994	2928	515 (18)	1300 (44)	1109 (38)	4 (0)	0 (0)	0.78
		1995	2928	469 (16)	1380 (47)	1061 (36)	18 (1)	0 (0)	0.79
T1-1									
	T1-1 (Lower)	1993	1859	265 (14)	1127 (61)	441 (24)	26 (1)	0 (0)	0.84
		1994	2928	536 (18)	1375 (47)	858 (29)	157 (5)	2 (0)	0.78
		1995	2928	626 (21)	1512 (52)	724 (25)	66 (2)	0 (0)	0.80
	T1-1 (Upper)	1993	-	-	-	-	-	-	-
		1994	813	211 (26)	391 (48)	74 (9)	137 (17)	0 (0)	0.69
		1995	2928	839 (29)	1600 (55)	489 (17)	0 (0)	0 (0)	0.82
T1-2									
	T1-2 (Lower)	1993	1912	358 (19)	679 (36)	830 (43)	28 (1)	17 (1)	0.75
		1994	2928	677 (23)	1710 (58)	541 (18)	0 (0)	0 (0)	0.82
		1995	2928	802 (27)	1681 (57)	445 (15)	0 (0)	0 (0)	0.82
	T1-2 (Upper)	1993	1906	131 (7)	1059 (56)	620 (33)	96 (5)	0 (0)	0.81
		1994	2928	450 (15)	1156 (39)	1153 (39)	162 (6)	7 (0)	0.73
		1995	2928	551 (19)	1247 (43)	904 (31)	205 (7)	21 (1)	0.73
T1-3 (Lower)									
	T1-3L (Lower)	1993	1855	218 (12)	1091 (59)	502 (27)	44 (2)	0 (0)	0.85
		1994	2928	517 (18)	1464 (50)	913 (31)	34 (1)	0 (0;	0.79
		1995	2928	609 (21)	1456 (50)	771 (26)	92 (3)	0 (0)	0.79
	T1-3L (Upper)	1993	1908	114 (6)	1033 (54)	670 (35)	91 (5)	0 (0)	0.81
		1994	2928	544 (19)	1387 (47)	972 (33)	25 (1)	0 (0)	0.79
		1995	2928	529 (18)	1418 (48)	896 (31)	85 (3)	0 (0)	0.79
T1-3 (Upper)									
	T1-3U (Lower)	1993	1904	88 (5)	1015 (53)	692 (36)	109 (6)	0 (0)	0.79
		1994	2928	1123 (39)	1442 (49)	363 (12)	0 (0)	0 (0)	0.76
		19955	2928	1349 (46)	1210 (41)	342 (12)	27 (1)	0 (0)	0.72
	T1-3U (Upper)	1993	-	-	-	-	-	-	-
		1994	665	56 (8)	234 (35)	250 (38)	77 (12)	44 (7)	0.68
		1995	2928	532 (18)	1445 (49)	868 (30)	79 (3)	4 (0)	0.78

[a]after Jirka and Homa 1990.

bodies in the watershed (Jim's Lake, between T1-3U and T1-3L; and Copper Lake, between T1-3L and the outlet T1), presumably after lake surface waters have warmed and possible stratification has occurred.

Changes in daily temperature (both mean and maximum) through reaches T1-1 (treatment) and T1-2 (unharvested control) in the summer months of 1995 are shown in Figure 5. In the control stream (T1-2), daily mean and maximum temperatures were reduced through the reach. The change in daily mean temperatures in T1-2 varied from -5.1 to 1.4°C ($\bar{x}$= -1.8°C) and daily maximum temperatures from -6.9 to 2.4°C ($\bar{x}$ =- 2.8°C). Conversely, daily mean and maximum temperatures increased through the treatment reach (T1- 1). The change in daily mean temperatures in T1-1 varied from -0.9 to 2.7°C ($\bar{x}$ = 0.9°C), and daily maximum temperatures from -1.1 to 7.2°C ($\bar{x}$ =1.6°C).

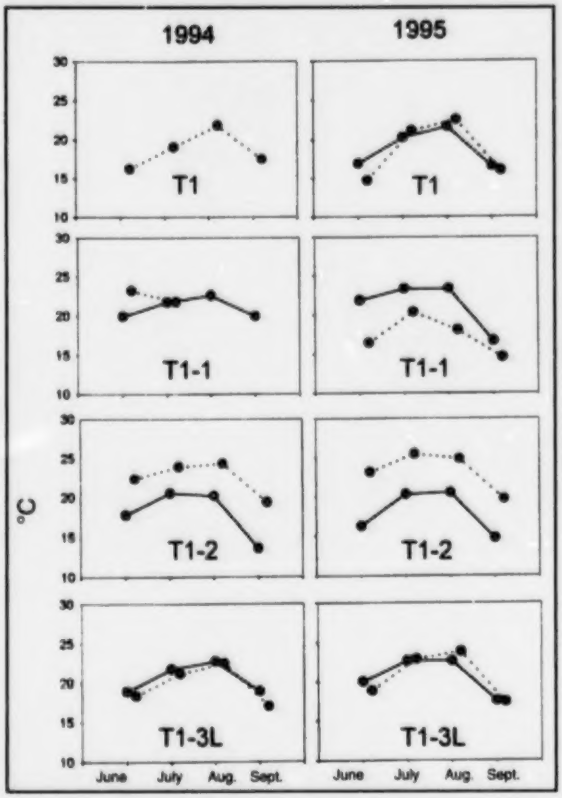

Figure 3. Maximum summer monthly temperatures in the upper (dotted line) and lower (solid line) stations for each of the five study reaches in 1994 and 1995.

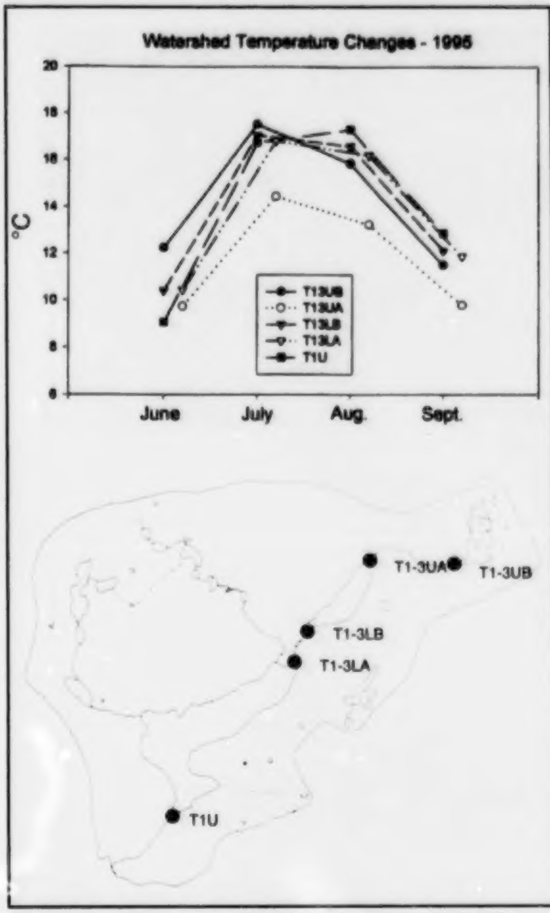

Figure 4. Mean monthly summer temperature dynamics though the watershed in 1995 at five thermograph locations.

Changes in diel temperature fluctuations through reaches T1-1 (treatment) and T1-2 (unharvested control) in the summer months of 1995 are illustrated in Figure 6. In the control reach (T1-2), the diel fluctuation at the upper end of the reach varied from 0.8 to 10.9°C ($\bar{x}$ = 4.6°C) and this diel variation was reduced over the reach ranging from 0.6 to 10.8°C ($\bar{x}$ = 2.6°C) at the lower end. Again, the opposite trend was evident in the treatment reach (T1-1). The diel fluctuation at the upper end of the reach ranged from 0.1 to 9.8°C ($\bar{x}$ = 2.4°C) and the variation increased over the reach from 0.2 to 11.3°C ($\bar{x}$ = 3.9°C) at the lower end.

Daily temperature regimes (mean, minima, and maxima) for the upper and lower stations in the treatments reach (T1-1), in relation to the five temperature strata defined for brook trout, are contrasted in Figure 7. At the upper station, daily temperature fluctuations were moderate, and temperatures (hourly) were within the optimum range 55% of the time (Table 2). Temperatures varied into the upper range (489 hours, 17% of the time); however, temperatures never reached stress levels. At the lower end of the reach, daily temperature variations increased; however, temperatures (hourly) remained in the optimum range 52% of the time (Table 2). Temperatures were in the upper range for 724 hours (25% of the period) and temperatures ranged into the stress levels (66 hours, 2% of the period) (Table 2).

Summer thermal habitat suitability indices (HSIs) were calculated for all sites for all years (Table 2). For locations with a complete record, HSIs were generally high and varied from 0.72 to 0.82. The changes in HSI over the stream reaches were relatively minor. The most appreciable difference occurred in T1-3

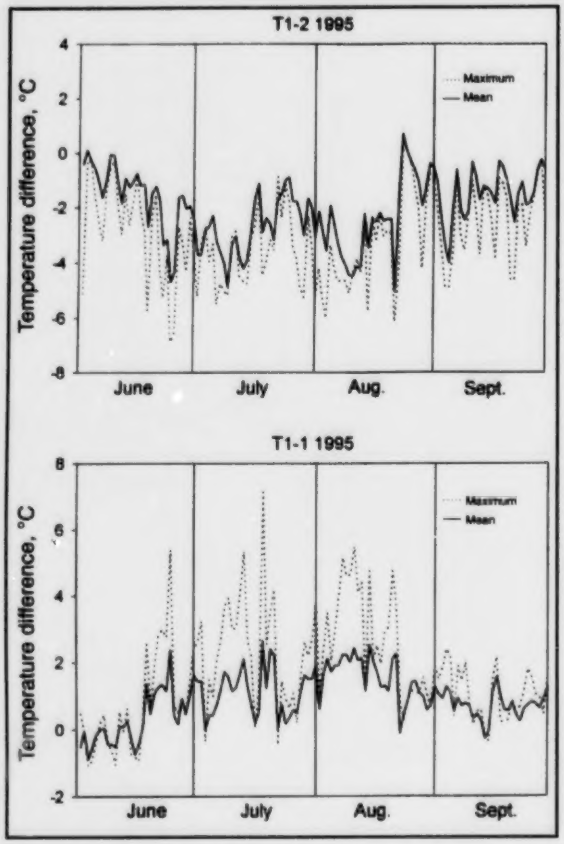

Figure 5. Summer daily temperature (mean and maximum) differences over the treatment (T1-1) and control (T1-2) stream reaches in 1995.

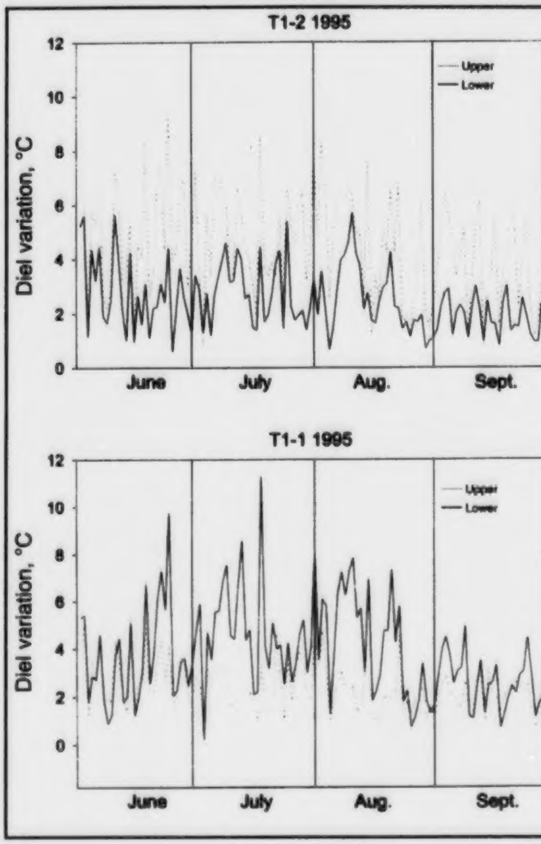

Figure 6. Summer diel temperatures fluctuations over the treatment (T1-1) and control (T1-2) stream reaches in 1995.

Upper in 1995. This decrease (from 0.78 to 0.72) was related to the cooling effect over the reach with a greater proportion of hours with temperatures below the optimum range (1349 hours, 45% of the period at the lower station). Changes in HSI associated with harvesting in T1-1 in 1995 were also minor (a decline from 0.82 to 0.80).

Discussion

The dynamics of temperature variation and regulation through the unperturbed fluvial and standing waters in the Copper Lake Watershed provide an indication of the role of small streams and associated riparian vegetation in maintaining the thermal conditions associated with headwater areas. As water flows downstream, water temperature tends to equilibrate with air temperature, a process influenced by local environmental factors such as stream shading, wind, humidity, groundwater influence, and standing water effect (Beschta et al. 1987). The most important hydrological factors affecting temperature regulation are the source of water (i.e., surface or subsurface lake outlet, snowmelt, groundwater), stream volume and rate of flow (Ward 1985). Temperature changes are greatest in summer months when air temperatures are highest, days are longer and sun angles are higher, and stream discharge is at seasonal lows (Beschta et al. 1987). In the unperturbed stream reaches, it is apparent that these small streams with good streamside and canopy cover were cooled along their length in the summers of both 1994 and 1995. Mean monthly temperature decreases over these reaches were as large as 3.1°C (T1-3 Upper, July 1995). These headwater fluvial systems are thereby maintained cool and stable, and consequently are valuable rearing areas for resident salmonids.

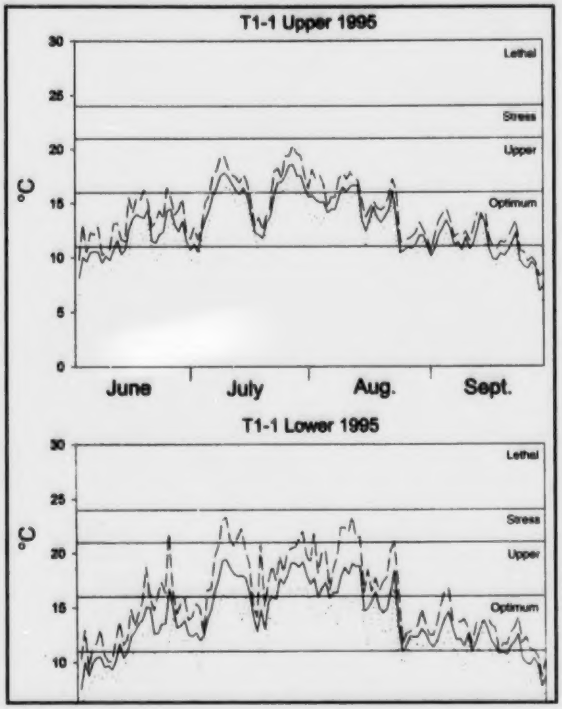

Figure 7. Daily summer temperature regime (mean, minimum, maximum) for the upper and lower stations on the treatment reach (T1-1) in 1995, in relation to temperature strata defined for brook trout.

Standing waters in the drainage basin influenced temperature variation in 1995. During the summer months in the upper watershed (T1-3), a warming trend was evident from the upper to lower reaches associated with passage through Jim's Lake. This was largely a result of the extensive cooling in the upper reach as stream temperatures are lowest within the entire watershed at the lower end of T1-3 Upper. Passage through Copper Lake, the largest standing water body in the watershed, maintained the overall cooling trend in June, while July temperatures were similar, followed by warming in the late summer months of August and September. This was likely due in part to the temperature dynamics within Copper Lake itself. Copper Lake was ice covered into mid-May and waters were well mixed and cool in the early summer. As the summer progressed, the surface waters warmed and, because of the large lake volume, it is probable that the lake stratified in the late summer. This was reflected in the shift from cooling to warming of water temperatures through Copper Lake as the summer progressed. These dynamics will be important to assess the effect of harvesting in headwater systems on downstream reaches.

Despite the relatively small amount (1.82 ha) of harvesting in T1-1 in the fall of 1994, albeit without a buffer strip, there was a major shift in the temperature dynamics associated with this reach as compared to unperturbed fluvial habitats in the watershed. The removal of riparian canopy can result both in higher summer water temperatures and greater daily fluctuations (Ringler and Hall 1975). Changes in average maximum temperature following removal of forest canopy in North American locales have generally ranged from 3 to 10°C (Beschta et al. 1987). Increased mean, maximum and diel variation in temperatures were apparent on the harvested reach while the opposite was evident in the unharvested streams. These dynamics may also, in part, be influenced by the small lake at the top of the reach.

Harvesting can also increase the daily extent and amount of time that high temperatures exist, thus increasing stress (Lynch et al. 1984). The increased exposure of small streams associated with clearcuts has been shown to result in substantial increases in the diel fluctuations (Beschta et al. 1987). Lynch et al. (1984) reported large, rapid diel fluctuations (up to 17°C) associated with clearcutting without a buffer strip and speculated that these changes were sufficient to thermally stress brook trout. Lynch et al. (1984) also reported that harvesting without a buffer strip increased the period during which temperatures were in the stress range (> 21°C) for brook trout, while slight temperature increases associated with a clearcut with a buffer strip resulted in optimum temperatures for brook trout more frequently. These changes were also evident in T1-1 where there was an increase in the number of hours at stress levels (> 21°C) and in diel variation, which could indicate an increase in overall stress conditions for brook trout as a result of forest harvesting.

Swift and Messer (1971) suggested that the effects of clearcutting on stream temperatures would be greatest in small nursery streams. Heavily canopied and spring-fed tributary streams tend to be much cooler than higher order streams (Ward 1985). Shading effects from riparian vegetation greatly moderate and reduce energy exchanges at the water surface (Beschta et al. 1987). Small streams respond rapidly to exposure to solar radiation because they lack the volume and flow rate to dissipate thermal energy. For these reasons, small streams exposed to direct solar radiation and braided channels tend to

exhibit the greatest diel ranges in water temperature, and these ranges are greatest in the summer (Ward 1985). This highlights the important role of fluvial habitats in thermo-regulation (cooling) of the summer water temperatures in this watershed, as well as the susceptibility of lotic habitats to temperature effects of forest harvesting.

Swift and Messer (1971) demonstrated that small-stream temperature changes associated with clearcuts can be ameliorated to a degree by buffer strips. In most instances, retention of the riparian vegetation along streams prevents unacceptable increases in water temperature (Binkley and Brown 1993). It was determined that temperature increases associated with clearcuts with a riparian leave strip are a result of increased soil temperatures influencing the temperature of runoff and increased groundwater temperatures (Brown and Krygier 1970). Holtby and Newcombe (1982) determined that summer temperature increases were proportional to the basin area harvested.

The change from a cooling to a warming effect associated with harvesting in these small headwater systems may lead to cumulative temperature increases in downstream reaches. Maximum temperatures typically demonstrate a gradual increase from the headwaters to the river mouth (Ward 1985). The extent of downstream changes depends on the temperatures of the contributing tributaries weighted by their respective discharges (Beschta et al. 1987). These cumulative increases could adversely affect downstream temperatures and lead to displacement by creating lethal or unsuitable conditions. In areas supporting salmonids, these cumulative increases may be sufficient to decrease the quality of downstream habitats. This decrease in quality can be further exacerbated as stream order increases, as stream width increases exposure to direct solar radiation, causing an altered temperature regime for some distance downstream (Ward 1985).

Temperature increases can lead to physiological stress inducing fish to move to seek thermal refugia resulting in habitat abandonment (Swift and Messer 1971). Lynch et al. (1984) observed that temperature changes related to forest harvesting may be sufficient for brook trout to migrate from the affected stream. Despite the fact that temperatures are unsuitable for only a period of time, these areas can be underpopulated and lower fish production because the fish that leave the reach may not necessarily return. This may be particularly important in small nursery streams, usually populated by juvenile fishes, such as in the Copper Lake Watershed, where the majority of fish in the fluvial habitats were underyearling and yearling brook trout (Scruton et al. 1995). These age classes of trout are not thought to migrate as readily as older age groups (personal communication, J. Gibson, Fisheries and Oceans, St. John's, Newfoundland). Post-harvesting migration studies on T1-1 in 1995 have demonstrated the abandonment of habitat as fish have revealed a net movement from the stream to the lake (McCarthy et al. 1998). These movements, however, may not be associated only with temperature changes; they may be closely linked to sedimentation events (Clarke et al. 1998).

An important aspect to consider in addition to the extent (amount) of harvesting related alteration of temperature regimens is the duration of these changes. The duration of temperature effects from harvesting without a buffer strip will depend on the ability of the watershed to redevelop altered riparian (i.e., shade) conditions. Riparian shrubs and grasses would quickly recover; however, it could take considerably longer (40 years or more in Newfoundland) for the canopy to redevelop (Newfoundland Forest Service 1994). Small streams may benefit from the rapid regrowth of riparian vegetation more than larger rivers, which would require the re-establishment of the larger overstory canopy cover for the shade benefit to be established. This will be an important consideration in the overall assessment of forest harvesting effects in Newfoundland and the ability of riparian leave strips to ameliorate these effects.

Conclusions

The temperature characteristics of the unperturbed fluvial habitats in the Copper Lake Watershed indicated that the habitats are high quality in thermal terms, and temperatures were within preference and tolerance ranges for brook trout for a majority of the summer period. Warming of streams to stress or potentially lethal levels occurred infrequently. Unharvested fluvial reaches provided a cooling effect throughout their lengths over summer months. The standing waters in the watershed (Copper Lake and Jim's Lake) result in a warming effect as the summer progressed. Presumably this was related to increased temperatures in the surface waters and possible stratification.

A summer thermal habitat suitability index (HSI) was calculated for the study reaches to provide a convenient index to describe the thermal habitat quality. This index indicated that the streams in the Copper Lake Watershed were thermally suitable for

brook trout and that the limited harvesting that has occurred has not significantly altered the thermal HSI. This will be studied in future investigations examining the relationship between forest harvesting and stream temperature regimes integrating fisheries research with other biological studies (e.g., wildlife and avian studies) in the Western Newfoundland Model Forest.

The initial effects of limited harvesting on one of the study streams has apparently altered the temperature dynamics over the reach. Daily mean, maxima, and diel variation have increased in 1995 in the treatment while the opposite dynamics were apparent in an unperturbed stream. Despite these changes, the temperature conditions in the treatment reach still indicate high quality thermal brook trout habitat. However, the shift from a cooling to a warming trend in these fluvial habitats after harvesting suggests potential concerns related to cumulative warming in downstream reaches.

The provision of buffer or leave strips along stream margins is the single most important step that can be taken to ameliorate the potentially harmful effects of forestry practices on fish habitat (Barton et al. 1985, Murphy et al. 1986, Ontario Ministry of Natural Resources 1988). Streamside vegetation plays an important role in regulating water temperatures, and shading from riparian vegetation can be the primary governing factor in determining the survival of a salmonid populations in streams/rivers with borderline summer temperatures.

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Large Woody Debris Dynamics and its Relation to Juvenile Brook Trout *(Salvelinus fontinalis)* Densities in Four Small Boreal Forest Headwater Streams of Newfoundland, Canada

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Abstract

The large woody debris (LWD) dynamics of four small boreal forest headwater streams in the Copper Lake watershed, Newfoundland, Canada was monitored during 1994 and 1995. Frequency and volume of LWD in the stream channel did not show any trends related to stream size but submerged volume increased with stream size. The small primary streams had more LWD lying perpendicular in the stream channel (32–53%) than did the secondary streams (24–26%), and orientation characteristics were similar over the 2 years. Large woody debris volume was stable in the smaller primary streams over the 2-year period while frequency changed significantly. The exception to this observation was the loss of LWD from a stream that had 20% of its length clearcut to the water's edge (i.e., no riparian buffer strip) between surveys. This stream was observed to have both significant decreases in LWD frequency and volume, which could be attributable to the loss of source material (forest harvesting) coupled with the flushing of LWD from the stream channel during a major rainstorm event. Large woody debris was negatively correlated to brook trout densities bringing into question the role LWD plays in juvenile salmonid production in these small boreal forest streams. Study streams were populated primarily by under-yearling and yearling trout. This population pattern may be related to the microhabitat conditions associated with LWD and the habitat preferences of the age (size) classes. Additionally, the distribution of LWD, in consideration of the high gradient of these streams, may play a role in limiting the distribution of juvenile fish from spawning/incubation locations.

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Introduction

The role of large woody debris (LWD) in salmonid streams has been studied extensively over the past 20 years. These studies have outlined the importance of LWD in the retention of sediment and inorganic particles (Bilby and Likens 1980; Bilby 1981; Ward and Aumen 1986; Aumen et al. 1990) and the creation of stream channel features such as microhabitats, cover, and pools (Andrus et al. 1988; Robinson and Beschta 1990a; Carlson et al. 1990; Gregory and Davis 1991; Ralph et al. 1993). The net effect of these attributes is to increase the sinuosity and complexity of the stream while reducing the harmful effects of sedimentation and nutrient loss, making the stream more productive for salmonids. It is therefore not surprising that numerous studies have linked salmonid density and/or biomass with the complexity of LWD in the stream channel (e.g., Bryant 1983; Elliott 1986; Lisle 1986; Faush and Northcote 1991; Flebbe and Dolloff 1995). Recently, research has focused on characterizing the physical aspect of LWD (i.e., size, volume, and orientation) and its relation to stream size and optimums for fish habitat characteristics (Bilby and Ward 1989, 1991; Robinson and Beschta 1990b; Fausch and Northcote 1991).

Most LWD studies have been conducted in the Pacific Northwest (e.g., Lisle 1986; Elliott 1986; Ward and Aumen 1986; Andrus et al. 1988; Bilby and Ward 1989, 1991; Robinson and Beschta 1990a, 1990b; Aumen et al. 1990; Carlson et al. 1990; Fausch and Northcote 1991; Ralph et al. 1993) with a few originating in the eastern United States (e.g., Bilby and Likens 1980; Bilby 1981; Flebbe and Dolloff 1995). The bio-physical conditions, climate, forest, and fish fauna of these areas are different from the boreal forest of eastern Canada; therefore, results cannot be readily transferred. The present study was conducted in the boreal forest of Atlantic Canada where there is a paucity of published material on this subject. The boreal forest of Atlantic Canada generally has smaller trees providing LWD to the streams, and the streams themselves are generally small. These differences are most pronounced in insular Newfoundland, where the forest is primarily composed of small coniferous trees [e.g., black spruce (*Picea mariana*) and balsam fir (*Abies balsamea*)]. Many important salmonid nursery streams are associated with small headwater systems, where streams are generally less than 10 metres wide. Additionally, insular Newfoundland is characterized by a sparse fish fauna, dominated by salmonids (Scott and Crossman 1964).

Most of Newfoundland's harvestable wood lies within these small headwater watersheds; consequently, the potential for interactions between forestry and fishery resources is high. Recently, a large multi-agency, multi-disciplinary research study (Copper Lake Buffer Zone Study) has been initiated under the auspices of the Western Newfoundland Model Forest (Scruton et al. 1995). The main purpose of this study is to conduct region-specific research on the benefits of providing unharvested buffer strips in riparian zones for the protection of fish and wildlife resources. An important component of this research program has included LWD studies, because this is a natural dynamic of forest succession and the quality of fluvial fish habitat can be affected by forest management practices (e.g., harvesting). Study objectives included characterization of the LWD dynamics of a number of small streams in a headwater watershed and investigation of the relationship between brook trout density and LWD in these streams. Additionally, the initial effects of a partial clear-cut, without a buffer zone, on the LWD dynamics of one of these small streams has been investigated.

The Study Area

The Copper Lake watershed is a small headwater system (13.5 km^2) located about 17 km southeast of Corner Brook, Newfoundland, Canada (Fig. 1). The watershed is located 350–650 metres above sea level and has an average annual rainfall of 1186 mm. Streams in the watershed have moderate to high gradients ranging from 2.5 to 23.8%. Soils are predominantly moderate to coarse glacial tills derived from intensely deformed and highly metamorphosed rocks (Kennedy 1981), which have a relatively large moisture content. These soil characteristics, coupled with the steep hillside slopes, create an increased potential for erosion (van Kesteren 1992). The shallow nature of the soils can create opportunities for blow-down which can also increase erosion/sedimentation problems and input of LWD.

The vegetation of the area is typical of the Western Newfoundland Ecoregion (Damman 1983) and is largely composed of mature and over-mature balsam fir with interspersed black spruce and white birch (Betula lutea). Details on the study area as well as a description of the Copper Lake Buffer Zone Study are provided in Scruton et al. (1995).

Fluvial fish habitats in the watershed consist of five primary (1°) tributaries, of which three have been studied intensively (T1-1, T1-2, and T1-3), and the Copper Lake outlet (T1), a second order (2°)

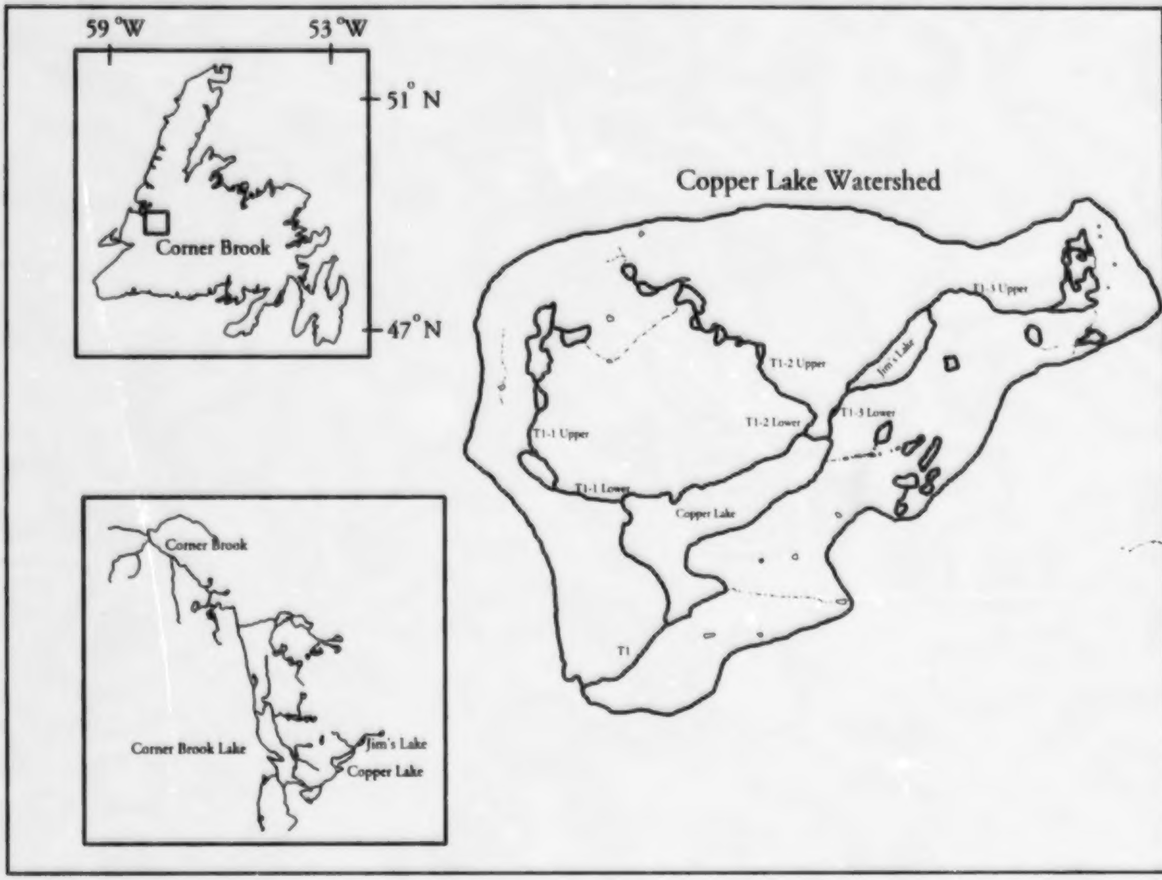

Figure 1. Location of the Copper Lake watershed with study streams highlighted.

stream. These streams are dominated by riffle habitats, with some occurrence of steady, run, and rapids with large pool habitat being completely absent. Forest canopy cover was extensive in the primary streams. Study streams have a number of large and small water falls, some of which are obstructions to upstream movement; debris jams are present on two of the primary tributary streams.

Materials and Methods

Studies have included comprehensive LWD surveys and brook trout population estimates on four small streams in the Copper Lake watershed during August 1994 and 1995. Evaluation of LWD dynamics was conducted through surveys of four stream reaches, including three primary (T1-1, T1-3 Lower and T1-3 Upper) and one second order stream (T1) (Fig. 1). Each survey consisted of walking the entire stream length and recording 1) location along the stream reach; 2) orientation to stream flow; and 3) the height above the water of all LWD greater than 8 cm diameter at the base, found within the stream channel. Total and submerged lengths of each LWD were recorded to facilitate the calculation of total and submerged volume. Volume was calculated assuming that a tree trunk roughly approximates a cylinder [$V = 0.33(\pi r^2 h)$] where r was the radius at the base of the tree and h was its total length or height. Volumes and numbers are presented per 30 m of stream length to aid in comparisons between streams and years. Statistical comparisons between the 2 years were conducted by averaging volume and numbers per stream section (sections were about 100 m long) while 95% confidence intervals around these averages were then calculated using randomization techniques (Edgington 1987).

Electrofishing surveys were used to obtain population estimates for brook trout (*Salvelinus fontinalis*) at three stations per stream reach immediately prior to the LWD survey to evaluate brook trout utilization

Table 1. Physical characteristics of the streams used in the Large Woody Debris (LWD) survey

Stream	Stream order	Survey distance (m)	Average wetted width during survey (m)	Gradient (%)	Mean depth (cm)	Average canopy cover (%)	Disturbance level
T1	2	1399	5.67	5	14.15	27.7	Road crossing (Arch culvert)
T1-1	1	527.8	2.61	6	6.5	45	Upper 200 m cut to stream's edge and road crossing (culvert)
T1-3 Lower	1	415.3	4.93	2.5	10.88	71.7	Undisturbed
T1-3 Upper	1	1257	2.61	6.7	8.28	15.8	Undisturbed

of LWD in these small streams. Brook trout population estimates were conducted using a depletion method as described by Scruton and Gibson (1995). Large woody debris levels in each electrofishing station were correlated with population densities to explore relationships between the two variables. The best method of portraying LWD as a predictor of brook trout population estimation was then regressed against population density values to develop significance (p-values) and explained variance (r^2) levels.

Results and Discussion

The study streams within the Copper Lake watershed are small, moderately sloped, primary (n = 3) and secondary streams (n = 1), ranging in length from 415 to 1400 metres, average wetted width varying from 2.6 to 5.7 meters, and gradients of from 2.5 to 6.7% (Table 1). Streams have undergone varying degrees of disturbance (resource road construction, forest harvesting) since initiation of research in 1993 (Table 1) and are primarily used by brook trout for spawning and rearing of the young-of-the-year (YOY) and 1+ year classes. These small streams are unique in the analysis of LWD as it relates to salmonid production and forestry practices, because most of the published work in this area has originated from western North America and the eastern United States where streams often tend to be larger and the biological assemblages very different from those in the present study.

The frequency of LWD (no./30 m) averaged over the 2-year period was highest in the 2° stream T1 followed by T1-3 Upper, T1-1, and T1-3 Lower, respectively (Fig. 2). There was no trend observed between stream size and frequency of LWD as has been the case with studies conducted on the west coast of North America (Bilby and Ward 1989; Robinson and Beschta 1990b; Bilby and Ward 1991),

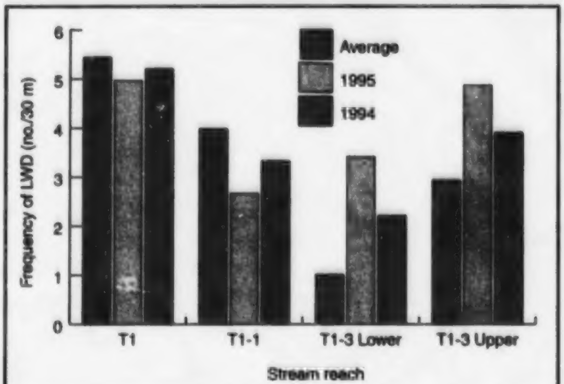

Figure 2. Frequency of LWD (No./30 m) in the four study streams during 1994 and 1995.

and the observation of the highest frequency in the largest stream directly contradicts the findings of these studies. The lack of a relationship between LWD and stream size may, in part, be a function of the narrow range of widths in the study streams.

Changes in LWD frequency between 1994 to 1995 were most pronounced in T1-3 Lower and T1-3 Upper with a 3.4- and 1.7-fold increase, respectively (Fig. 2), both of which were significant ($p < 0.05$). In contrast, T1-1, the other 1° stream, had a significant ($p < 0.05$) 1.5 fold decrease in the numbers of LWD from 1994 to 1995 (Fig. 2). A decrease in numbers was also observed in the 2° stream (T1); however, the 1.9-fold change was not significant ($p < 0.05$) (Fig. 2). An increase in LWD number from 1994 to 1995 appears to be a natural trend for the smaller streams in the watershed. This increase was likely related to a severe rainstorm in the watershed in early June of 1995. Estimated discharge at T1-1 Lower (J.H. McCarthy, Memorial University of Newfoundland, unpublished data) was 3584 m^3s^1, or about 896 times

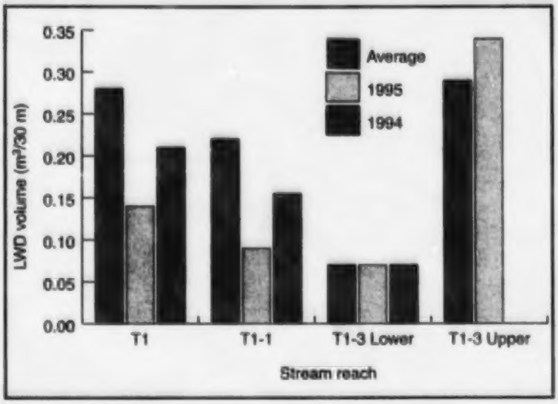

Figure 3. LWD volume (m^3/30 m) in the study streams of the Copper Lake watershed during 1994 and 1995.

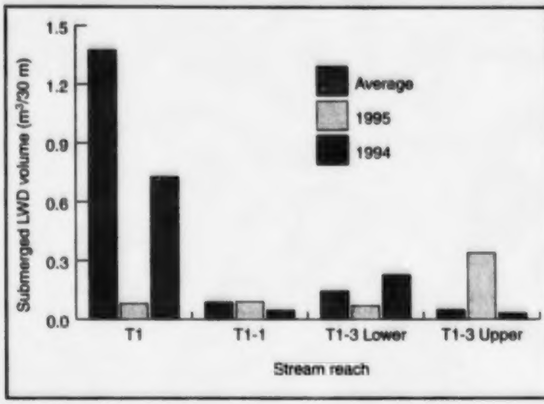

Figure 4. Submerged LWD volume (m^3/30 m) in the study streams of the Copper Lake watershed during 1994 and 1995.

the mean annual discharge, and approximately a 176-fold increase in seasonal discharge levels before this storm event. The difference observed between the other 1° streams, T1-3 Upper and T1-3 Lower, and T1-1 was most likely due to harvesting of the top 200 m, without a riparian buffer strip. This removal of trees meant that T1-1 did not have a source of LWD to draw from in 1995, and the high water levels caused by the rainstorm reduced the frequency of LWD by flushing the channel.

T1-3 Upper had the highest observed volume of LWD per 30 m of stream in both 1994 and 1995 followed by T1, T1-1, and T1-3 Lower, respectively (Fig. 3). There was no discernible trend between LWD volume in the stream channel and stream size. The observation of a small 1° stream having a larger volume than a 2° stream directly contradicts trends observed in western North American streams (Bilby and Ward 1989; Robinson and Beschta 1990b; Bilby and Ward 1991) but only one 2° stream was surveyed in this study.

The LWD volume per 30 m of stream between 1994 and 1995 decreased 2.0- and 2.4-fold in T1 and T1-1, respectively (Fig. 3), both of which were significant ($p < 0.05$). These decreases correspond to decreases in number of LWD observed in these streams over the same period. Volumes in T1-3 Upper and T1-3 Lower did not change significantly from 1994 to 1995; however, T1-3 Upper demonstrated an increase in volume corresponding to an increase in number. The only stream to have differing trends in numbers and volume from 1994 to 1995, T1-3 Lower, demonstrated an increase in numbers (Fig. 2) while volume remained the same (Fig. 3). These observations suggest that the larger LWDs observed in T1-3 Lower in 1994 were flushed from the stream channel, presumably during the rainstorm of June 1995, and replaced by smaller LWDs with greater frequency in 1995. Consequently, all streams except T1-3 Upper, were observed to have a pronounced flushing of LWD from the stream channel due to the hydrological peaks related to the June 1995 rainstorm.

Observations of submerged LWD volume did follow a continuum based on stream size where the widest stream was observed to have the most submerged LWD on average with the smaller streams having less submerged LWD (Fig. 4). All streams in the Copper Lake watershed were observed to have a reduction in the volume of submerged LWD from 1994 to 1995, with the exception of T1-3 lower. The reductions in submerged volume observed in T1 and T1-1 were 17.0-fold and 14.3-fold, respectively, indicative of relative rates of flushing of LWD observed in these streams due to the high water flows in June 1995 and the forest harvesting in T1-1.

The orientation of LWD within the streams was calculated on a percentage basis to discern any trends within the watershed and to make a preliminary comparison of these trends with observations from the west coast of North America. Smaller 1° streams had more LWD placed perpendicular to the stream flow (32–53%) than did the 2° stream T1 (24–26%) (Fig. 5), and this is likely related to relative difference in stream energy and the ability to dislocate LWD during hydrological events. These observations are

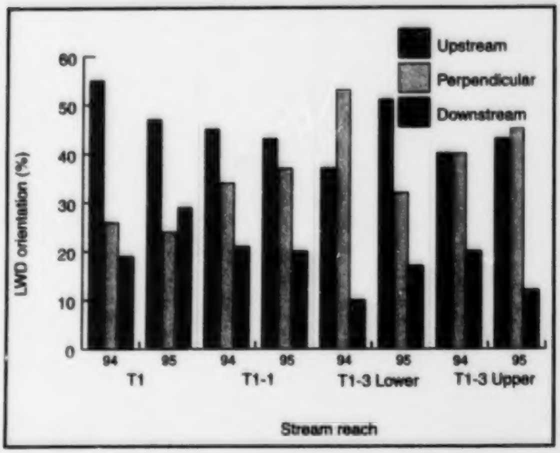

Figure 5. Orientation of LWD in the study streams of the Copper Lake watershed during 1994 and 1995.

Table 2. Correlation co-efficients (r) for LWD measurements and population estimates of brook trout in Copper Lake electrofishing stations

LWD measurement	Density (No./100 m^2)	YOY density) (No./100 m^2)
LWD no.	-0.46	-0.47
LWD vol.	-0.54	-0.50
LWD no./30 m	-0.42	-0.40
LWD vol./30 m	-0.51	-0.45

similar to those noted by Bilby and Ward (1989) in Washington streams but differ from observations in southeast Alaska by Robinson and Beschta (1990b) but cannot be viewed as representative for insular Newfoundland due to the small number of streams surveyed (n = 4). Little change in LWD orientation was observed from 1994 to 1995 except in T1-3 Lower where many perpendicular LWDs were replaced (or displaced) by/to LWDs directed downstream relative to the flow.

The relationship of LWD and brook trout use in these small streams was investigated by correlating four measures of LWD to estimated population density for total brook trout and YOY brook trout. All measurements of LWD and brook trout density were negatively correlated with r-values ranging from –0.40 to –0.54 with the best predictor of trout abundance being the total volume of LWD in the electrofishing station (Table 2). This relationship was significant ($p = 0.006$), and LWD volume in the electrofishing station could explain 29.5% of the variation in the data set. The negative relationship between brook trout density and LWD observed in these streams differs from observations made by Flebbe and Dolloff (1995) in streams in North Carolina and suggests that the mechanisms controlling brook trout density and habitat differ in the Newfoundland situation. One plausible explanation for this difference is related to effect of LWD, in addition to the high gradient in the study streams, in restricting distribution and movement of the younger age classes of trout to available habitats. Additionally, microhabitat conditions associated with LWD, primarily an increase in depth and cover, are more preferred by larger, older trout than the YOY and 1+ trout that primarily use these streams (Raleigh 1982). Further, the density of YOY trout would also be related to spawning escapement and reproductive success, which in turn could be affected by access and microhabitat conditions associated with LWD. These relationships suggest that the role of LWD in salmonid production in small boreal forest streams in Newfoundland is different from that reported in other studies, and further research is required to elucidate the mechanisms of density control in these environments.

Conclusions

Large woody debris characteristics of the streams in the Copper Lake watershed were both similar to and different from the characteristics reported for streams of the Pacific Northwest. Streams in the Copper Lake watershed did not have any discernible trends in frequency or volume of LWD in relation to stream size, although a relatively narrow range in stream size (width) was represented in the study watershed. Submerged volume and orientation characteristics were, however, similar to those reported by Bilby and Ward (1989).

Distinct differences in LWD characteristics were observed between the 2 years of the study. These differences were attributable to a flushing and replacement of LWD caused by the high discharges of June 1995. These observations suggest possible large inter-annual variation in LWD dynamics for this watershed. These variations stem from the flashy nature and the high gradients of the streams in this area, which would generally contribute to instability of some habitat features in the study streams.

The partial clearcut of T1-1 Lower resulted in significant decreases in LWD frequency and volume in this stream. These decreases were likely related to lack of available trees to provide and replace LWD, after flushing of the existing debris in June 1995. These observations are similar to those reported in other studies (e.g., Toews and Moore 1982) and gives some insight into how LWD dynamics in small streams may be affected when the streams are harvested to the water's edge. This observation would suggest that, in streams associated with harvesting without a riparian buffer strip, an expected effect would be the eventual loss of LWD and associated fluvial habitat characteristics, if hydrological events are sufficient to remove the material. This would also be compounded by the effects of clear-cutting on the hydrological regime, specifically the increase in frequency and magnitude of peak flow events (Dickison et al. 1981).

The relationship between LWD and brook trout density was both significant and negative in these streams. This finding is contrary to those from other studies and suggests that the mechanisms linking LWD to salmonid density may be different in these small streams. It is possible that the blockages to movement and distribution of YOY and 1+ trout related to LWD may be detrimental in these streams, due in part to their small size and steep gradient. The benefits of LWD and trout density may be related to the microhabitat conditions they create, with increased depth and cover associated with LWD preferred by larger, older trout, which are not abundant in these streams. The relationship between LWD and brook trout density differs from relationships identified in other studies, and will be the subject of more in-depth investigation as part of the Copper Lake Buffer Zone Study before we can fully understand the role of LWD in small boreal forest streams in Newfoundland.

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The Effect of Logging and Road Construction on Brook Trout Movement in the Copper Lake Watershed, Newfoundland, Canada.

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Abstract

Movements of tagged brook trout (*Salvelinus fontinalis*) were monitored during early June to mid-October in the Copper Lake watershed, Newfoundland, Canada in 1994 and 1995. This research, part of a large buffer-zone study, was undertaken to determine the effects of clear-cut harvesting and associated road construction on brook trout movement and habitat utilization. Movement was measured by recapture of individually tagged fish using counting fences, fyke nets, electrofishing, and angling. Analysis revealed that a clear-cut of 20% of the treatment stream's length cut to the water's edge did not alter the strength of association between tagged trout and their initial capture location. In addition, the movement index in the treatment stream was not different from that of a control stream in the same watershed. However, a comparison of the movement patterns, where direction of movement and habitat type were considered, displayed a significant increase in movement out of the fluvial habitat in the clear-cut stream to the lacustrine habitat. An increase in suspended sediment appears to be the major factor influencing this change in behavior. This modified approach to analysis, considering habitat type, direction of movement, and degree of movement, will be applied to assess the benefit of a 20-m buffer strip in ongoing studies.

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Introduction

The effects of forest harvesting practices on fluvial habitat pose serious implications for management of fisheries and forests in Newfoundland, Canada.

The effects of clear-cut logging on aquatic ecosystems in general have been documented since at least the mid-1930s (King 1937). The effects of forest harvesting practices on salmonids, however, are still not fully understood (Hicks et al. 1991). Additionally, much of our existing knowledge is based on research conducted in the Pacific Northwest and the northeastern United States and results are not readily transferrable to the boreal forests of insular NF. The Copper Lake Buffer Zone Study was undertaken in 1993 as an inter-disciplinary, multi-agency research effort to assess the impacts of forest harvesting practices in NF on riparian ecosystems (Scruton et al. 1995).

One objective of the study was to assess the impact of logging and road construction on fluvial and lacustrine habitats and changes in brook trout behavior, movement, and habitat use. Owing to their relatively high mobility and ability to avoid or exploit changes in their environment, fish can serve as initial indicators of changing conditions in aquatic habitats; furthermore, a knowledge of fish movements is often useful in identifying subtle changes which may not be readily detected by other means (Bergersen and Keefe 1976).

Materials and Methods

Study Site

The Copper Lake watershed is located approximately 17 km southeast of Corner Brook, Newfoundland, Canada 48° 49'17.5" N 57° 46'27.0" W) (Fig. 1). This area had not been previously harvested and contains a diversity of terrestrial and aquatic habitats (Scruton et al. 1995). The watershed was scheduled for harvest by Corner Brook Pulp and Paper Ltd. in 1994 and 1995.

Several streams within the watershed were surveyed for inclusion in the study. The two streams emphasized in this study included a control stream (T1-3) and a treatment stream (T1-1) (Fig. 1). The control stream was located in the northern part of the watershed where no forest harvesting or road construction was planned. It had an impassable waterfall 505 m upstream from its mouth. The treatment stream was in the southwestern part of the watershed where road construction and forest harvesting without buffer strips were planned. The treatment stream had an impassable waterfall 527 m upstream from its mouth. This study utilized the fluvial habitat below the impassable waterfall in each stream. Both streams had wetted widths averaging <2 m and were first-order, headwater streams (Scruton et al. 1995). Scruton et al. (1995) provides a detailed description of the area and study streams.

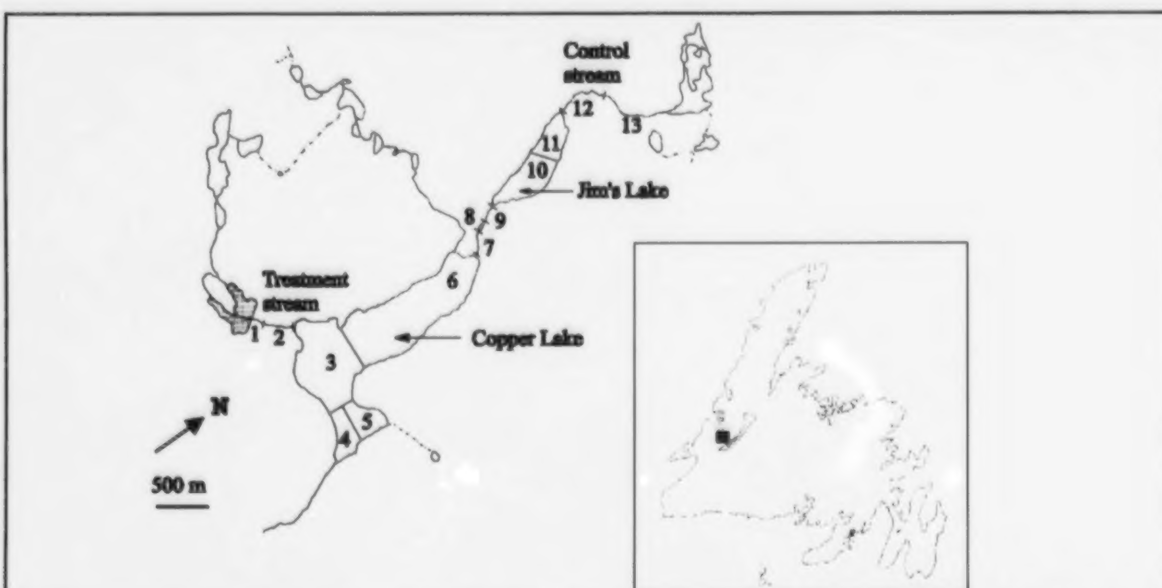

Figure 1. The location of the Copper Lake watershed, treatment (T1-1) and control (T1-3) streams, habitat sections, and clear-cut (shaded) within the watershed.

Road construction within the watershed began in the summer of 1994. In the fall of 1994, clear-cutting within the treatment stream (T1-1) drainage basin occurred and no buffer strips were left along the stream edge. This cutting schedule allowed the collection of 1 year of movement (1994) and 2 years of population and habitat data (1993–1994) before the cut and one year (1995), to date, of post-harvesting data. By the winter of 1994, the treatment stream had a road crossing (with cylindrical culvert) 300 m upstream and approximately 20% of its length was clearcut. This clear-cut was approximately 1.82 ha, constituted 9.0% of the stream's drainage basin, and was located on the top 100 m of the stream, below the impassable waterfall (Fig. 1).

Movement of Trout

Brook trout were captured by angling with flies (barbless hooks), electrofishing, fyke netting, and by counting fences from June 11 to October 7 in both 1994 and 1995. Stream and lake habitat within the study area were divided into habitat sections, with stream-study sections being separated by counting fences (Fig. 1). Fences were continually in operation for the study period except July 24–27, 1994, when they were washed out due to a rainstorm. Trout from each of the sections were tagged upon initial capture with individually numbered, color-coded fingerling tags (Floy model # FTF-69) and released. All fish were anesthetized in benzocaine (40 mg/mL acetone) at a final concentration of 5 mL per 8 L of water before handling (Brown 1993).

Index of Movement

A statistical comparison of the strength of association between the initial capture and final recapture location was made after Bergersen and Keefe (1976). A sample index of movement (h) was calculated for tagged trout from both stream populations based on the capture/recapture data (Tables 1 and 2). The approximate test of significance for the comparison of the index of movements was conducted at $p = 0.05$. The strength of association of tagged trout with their initial capture location approaches a value of 1 when the association is strong. A value of $1/\sqrt{RC}$ would indicate little or no association, where R is the number of rows in the contingency table and C is the number of columns. Each table had three columns. Each fish had a chance of being recaptured in every habitat section; hence, there were 13 rows.

No 1995 recaptures of fish tagged in 1994 were used in the index calculation so that seasonal time intervals were comparable. No fish was entered into the contingency table more than once so that all recapture observations were independent; that is, only the final recapture location within the time interval was recorded. As a result, only tagged fish were used. Some fish passed through the counting fences without being tagged. However, they were generally smaller fish (fork length <60 mm) whose behavior may have been altered by the tag (Xiao 1994). Brook trout caught in counting fences were recorded as recaptures because information about previous location and present location were known, much like a mark and subsequent recapture.

In July of 1994, the counting fence separating Jim's Lake from the control stream was damaged. This allowed fish to move into the stream without being caught by the fence for approximately 3 to 4 days. This event coincided with the time when larger fish started moving into the stream prior to spawning. Electrofishing of the stream was conducted after the damage was repaired, and because no large fish were in the stream before the storm, an estimate of the number of fish which entered the first stream section was made. These larger fish were tagged during electrofishing so that subsequent movements could be monitored.

Direction of Movements

The index of association does not take into consideration direction of movement (Bergersen and Keefe 1976) and hence could potentially mask a change in directional behavior of movement. This potential change may be important if there is a difference in habitat type between upstream and downstream movements; that is, if there is a difference between moving within a stream and moving between a stream and a lake.

Stream and lake study sections were grouped by habitat type, consequently, investigations could be made on movement patterns between different habitats (Leclerc and Power 1980). Chi-square tests were used to compare movement patterns between years ($p = 0.05$).

Two habitat types were used in the chi-square tests: lake (lacustrine) habitats and stream (fluvial) habitats. Combining data from some study sections was necessary for statistical purposes; lake sections were combined, the stream component of the behavior category "stream –> lake" has both movement from the upper and lower stream sections, and tagged fish which were recaptured in their initial capture location within the streams were also combined into one behavior category (no movement within stream). Combining these sections, however, does not

Table 1. Movement matrix of tagged brook trout from the treatment stream in 1994 and 1995 (in parentheses)

Station of final recapture	Station of initial capture 1	2	3	Totals
1	2 (1)	3 (10)	(1)	5 (12)
2	31 (2)	(2)	2 (6)	33 (10)
3	4 (3)	9 (40)	(3)	13 (46)
4	1	(1)		1 (1)
5		(1)		(1)
6				
7	1			1
Totals	39 (6)	12 (54)	2 (10)	53 (70)

Table 2. Movement matrix of tagged brook trout from the control stream in 1994 and 1995 (in parentheses)

Station of final recapture	Station of initial capture 11	12	13	Totals
4		(1)		(1)
10			(1)	(1)
11	1 (4)	13 (38)		14 (42)
12	103 (98)	11 (4)	11 (7)	125 (109)
13	2 (36)	13 (11)	1	16 (47)
Totals	106 (138)	37 (54)	12 (8)	155 (200)

impede comparing the movement of fish between habitat types. Both statistical methods were applied to the recorded movement patterns of brook trout populations within the treatment and control streams to determine if trout behavior and use of habitat within the treatment stream changed after harvesting and road construction.

Habitat Variables

Habitat variables monitored during the study included: mean daily air temperature (0.1°C); rainfall ($mm \cdot d^{-1}$); mean daily water temperature (0.1°C); dissolved oxygen (0.1 ppm); discharge (0.01 $m^3 \cdot s^{-1}$); water velocity (0.01 $m \cdot s^{-1}$); water depth (0.5 cm); streambed sedimentation (0.1 mg), total suspended solids (0.01 $mg \cdot L^{-1}$), and invertebrate biomass (0.01 $g \cdot m^{-2}$) (McCarthy 1996; Clarke et al. 1998, 1996; Scruton 1996). Details of measurement techniques can be found in Scruton et al. (1995).

Results

Index of Movements

The movement of brook trout was recorded for both streams in 1994 and 1995 (Table 1 and 2). The tables do not include all 13 habitat sections; sections with no recaptures were omitted. The sample indexes of movement for the treatment and the control streams between years as well as between streams within years were not significantly different ($p > 0.05$). The lower limit of h was 0.160. The 95% confidence interval for the population index of movement (H) for each of the stream populations broadly overlapped (Table 3).

Movement Patterns

Only the treatment stream had a significant difference in the movement patterns of brook trout between years (Table 4). There was a decrease in the proportion of fish moving downstream from the upper to the lower section and an increase in downstream movement to Copper Lake. In the control stream, there was no significant change in movement patterns between years (Table 5).

Other Movements

The greatest relocation distance was approximately 4.5 km. A fish that was tagged in the control stream in 1994 was recaptured at the outflow of Copper lake (Station 4) in 1995.

Table 3. Summary of calculations and index of movement of trout samples (h) and 95% confidence interval for population index of movement (H)

Stream (year)	$\bar{w}$	$h = e^{\bar{w}}$	$\sigma^2_{\bar{w}}$	95% C.I. for H
Treatment (1994)	-0.520	0.594	0.379	0.699 – 0.505
Treatment (1995)	-0.691	0.501	0.417	0.581 – 0.432
Control (1994)	-0.472	0.624	0.442	0.714 – 0.546
Control (1995)	-0.565	0.568	0.405	0.619 – 0.522

Table 4. Observed movement and expected values from the $\chi2$ tests for the comparison of trout movement patterns from the treatment stream between 1994 and 1995

Movement pattern	Observations 1994	Expected 1994	Observations 1995	Expected 1995
Upper→lower stream section	31	14.57	2[a]	18.42
Lower→upper stream section	3	5.74	10	7.26
Stream→lake	15	26.50	45[b]	33.50
Lake→stream	2	3.97	7	5.03
No movement in stream	2	2.21	3	2.79
Totals	53	53	67	67
χ^2_{calc}				46.228[c]

[a] larger than expected decrease.
[b] larger than expected increase.
[c] significant difference in movement patterns.

$\chi^2_{0.05,4} = 9.488$

Discussion

The calculated values of h for each stream sample ranged between 0.501 and 0.624 (Table 3). This suggests that the strength of association for each stream was weak whether cutting occurred or not. This was probably an artifact of the counting fences. Initial captures in the fences were treated as recaptures due to the fact that information about previous and present location was provided. Because the fences were operational continuously and hence provided the majority of movement information, the proportion of fish recaptured moving from their initial capture locations was probably inflated. This would reduce the strength of association between a fish and its initial location.

Shetter (1968) stated that brook trout are essentially sedentary in a habitat that offers adequate cover, food, and spawning sites. The low index values may also be an indication that these streams do not provide all these requirements. These trout populations do appear to move, to some degree, between habitat types throughout the season (McCarthy 1996). Studies have shown that if habitat is suitable for brook trout within an area, then migration outside that area is limited (Shetter 1968; LeClerc and Power 1980). However, when the scale of environmental changes exceeds the individual's capacity to respond *in situ*, the general biological response to adversity—migration—can occur (Taylor and Taylor 1977; Bjornn 1971; Shirvell and Dungey 1983; Gagen et al. 1989; Thorpe 1994), if migration is possible.

Changes in trout habitat within streams after forest harvesting and road construction has been studied extensively (Hall and Lantz 1969; Burns 1972; Feller 1981; Murphy and Hall 1981; Hewlett and Forston 1982; Johnson et al. 1986). Everest and Harr (1982) and Grant et al. (1986) suggested that if the area logged is less than 25–30% of the drainage area, impacts on habitat and trout abundance may not be significant. However, although the treatment cut within this study was only 9.0% of the drainage

Table 5. Observed movement and expected values from the χ^2 tests for the comparison of trout movement patterns from the control stream between 1994 and 1995

Movement pattern	Observations 1994	Expected 1994	Observations 1995	Expected 1995
Upper→lower stream section	11	8.72	7	9.28
Lower→upper stream section	13	11.63	11	12.38
Stream→lake	14	11.14	9	11.86
Lake→stream	105	115.77	134	123.23
No movement in stream	12	7.75	4	8.25
Totals	155	155	165	165
χ^2calc				9.358[a]

[a] no significant difference in movement patterns. $\chi^2_{0.05,4} = 9.488$

basin, constituting 20% of the stream length, changes in behavior and habitat use were apparent.

A behavioral response in the brook trout population of the treatment stream was evident in an increase in the proportion of fish leaving the stream and entering the lake and a decrease in downstream movement from the upper section to the lower section (Table 4). The decrease in downstream movement from the upper stream section was probably the result of fewer fish in that section. Electrofishing surveys in 1993 and 1994 showed population estimates of 25 and 17 fish respectively in the first 100 m of the upper section of the streams in August (Scruton and Daya 1994; Clarke et al. 1996). In 1995, there were only seven fish in this same section of the stream (Scruton unpublished data). The fact that there were fewer fish in the upper section was possibly due to decreased winter survival (Hicks et al. 1991; Johnson et al. 1986) or movement downstream in the spring or winter before the fences were in place.

The counting fences were inoperable due to high water flows for only 3 to 4 days for the entire 1994 field season. The number of fish entering the control stream prior to spawning, while the lower fence was washed out, was estimated. An accurate estimate of the number of fish moving downstream during the same storm could not be obtained. Observations during the 1995 season suggest that very little downstream movement occurred during this time of the season even with similar streamflows (McCarthy 1996).

Fences on the treatment stream were also inoperable for a short time (1 to 2 days) during the same storm. An estimate of the number of fish which moved in or out of the stream at this time could not be made. Therefore, the number of fish moving between the treatment stream and the lake may be slightly low for 1994. However, the movement patterns between years were still significantly different ($p < 0.05$) even if the number of fish moving out to the lake was estimated to be the same in 1994 as in 1995. The increase in movement from the treatment stream to the lake in 1995 occurred throughout the entire season, suggesting that the change in behavior is associated with a season-long change in habitat (McCarthy 1996).

Some habitat changes attributed to forest harvesting from other studies include streamflow regimes (Crisp 1993), water temperatures (Gray and Edington 1969), dissolved oxygen levels (Hall and Lantz 1969) and changes in invertebrate biomass (Hicks et al. 1991). These factors were measured during the present study and only minimum daily water temperatures appeared to be significantly decreased by clear-cutting (McCarthy 1996). This apparent lack of effect may be due to the small percent of the catchment harvested. A detailed assessment of the hourly temperature regime in the treatment stream indicated some subtle changes in 1) numbers of hours at various temperature strata; 2) thermal habitat suitability index; and 3) diurnal variation that could be attributed to the clear-cut harvesting without a buffer strip (Scruton et al. 1998). However, temperatures within the treatment stream rarely reached those considered to be the stressful upper limit and never reached temperatures considered lethal to brook trout (McCarthy 1996). Factors associated with road construction may have had more of an impact within the present cutting regime.

The major effect of road construction and logging activities appeared to be a significant increase in stream sedimentation within the treatment stream (Clarke et al. 1998). Road crossings can lead to the input of fine sediments from road surfaces that can restrict upstream movement (Hicks et al. 1991), increase physiological stress, decrease feeding, and increase the susceptibility of trout to bacterial disease (Redding et al. 1987). Such sublethal stress and reduced performance capacity may increase avoidance behavior. Alabaster and Lloyd (1982) suggested that suspended sediment measurements of less than 25 $mg \cdot L^{-1}$ are ideal to maintain fisheries and 25–80 $mg \cdot L^{-1}$ are acceptable to maintain good to moderate fisheries. The mean suspended sediment levels found in the treatment stream after harvesting were 161 $mg \cdot L^{-1}$; double Alabaster and Lloyd's acceptable level (McCarthy 1996).

Movement did appear to be correlated with increased flow rates. However, increased flows seemed to facilitate rather than cause movement (McCarthy 1996). As suspended sediments are also correlated to increased stream flows, it appears that total suspended sediment may be the variable most affecting brook trout movement at this time.

It is important to stress that these conclusions are developed after only 2 years of detailed study. At this point, there is little opportunity to observe year-to-year variation in movement and habitat use. At present, conclusions are drawn from contrasting observations between the treatment and control streams. Additional study is required to elucidate temporal (between years) aspects to behavioral changes as well as to identify causal factors for observed changes (Elliott 1994). With the limited number of years monitored to date, this study is only able to assess immediate results, which may not be representative of longer time sequences (Hall and Knight 1981). Monitoring the changes in habitats of control and treatment streams and the effects on behavior and habitat use of trout over the coming years will help assess if this observed change is persistent or detrimental to the population.

Conclusions

Fish did not appear to change the distance they traveled but they did change their direction of movement and the habitat type they moved into after the treatment cut and associated road construction. Brook trout emigrated from fluvial habitats within the treatment stream to lacustrine habitats after harvesting. It appears that possible increased suspended sediment within the treatment stream and temperature changes may be associated with this change in behavior. Further study is required to determine if the observed change in migration behavior is detrimental and/or persistent.

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The Effect of Logging and Road Construction on Fine Sediment Accumulation in Streams of the Copper Lake Watershed, Newfoundland, Canada: Initial Observations

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Abstract

The accumulation of fine sediment was monitored by the Wesche sediment method in the Copper Lake watershed, Newfoundland, Canada from June 1993 to October 1995 to evaluate the effect of two independent road crossings and limited logging on fine sediment accumulation in small boreal forest streams. The road crossings were observed to significantly increase fine sediment accumulation in the streams and these increases have persisted till now. Two differing construction techniques were used in the stream crossings: a bottomless arch culvert and a whole cylindrical culvert. There were no discernible differences in sediment accumulation between the two techniques immediately after construction. A limited clear cut above the road on one of the streams further increased sediment accumulation. There was very little inter-annual variation in sediment accumulation over the 2 years despite large differences in hydrological regime during the spring freshet. The change in sediment accumulation in the perturbed streams was higher over the summer period than after the spring freshet. An initial analysis of benthic community structure and abundance (1994) was not able to discern any significant differences in the affected streams when compared to two control streams in the watershed, with the possible exception of a reduction in Plecoptera abundance in one of the affected streams. Young-of-the-year brook trout (*Salvelinus fontinalis*) density has significantly decreased in one of the affected streams while the density in the other stream fluctuated throughout the study. The effects on brook trout populations were magnified in areas near the sediment source.

Clarke, K.D., Scruton, D. A., and McCarthy, J.H. 1998. The effect of logging and road construction on fine sediment accumulation in streams of the Copper Lake watershed, Newfoundland, Canada: initial observations. Pages 353–360 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

High levels of fine sediment in streams have been shown to reduce egg and alevin survival (Alexander and Hansen 1986; Everest et al. 1987), cause stress in larger fish (Servizi and Martens 1992), and reduce the diversity and/or density of benthic macroinvertebrates (Lenat et al. 1981). The net result of any of these conditions or their combination is the reduction in overall productivity of the stream, thus making sedimentation one of the most important problems associated with forest harvesting. It is, therefore, no surprise that the sources of fine sediment in forested watersheds have been extensively studied. One of the most prominent sources of sediment is the construction and use of logging roads (Beschta 1978; Reid and Dunne 1984; Everest et al. 1987; Eaglin and Hubert 1993). Most of the studies citing logging roads as an important sediment source have been surveys of a logged area (Reid and Dunne 1984; Eaglin and Hubert 1993); only a few studies have actually followed sedimentation throughout the course of forest harvesting practices within a watershed (Beschta 1978; Hartman and Scrivener 1990; see also the review in Everest et al. 1987).

Most of our knowledge of forestry–fishery interactions and the role of sediment in North America has been developed in the Pacific Northwest (Beschta 1978; Reid and Dunne 1984; Bilby 1985; Everest et al. 1987). The physical and biological conditions of the Pacific Northwest differ greatly from those of the eastern Canadian boreal forest (Scruton et al. 1995), which generally has shallow, erodible soils and a lower species diversity. These characteristics, coupled with observations of very low benthic densities and secondary production in Newfoundland inland waters (Clarke 1995), create a sensitive climate for external perturbations.

Most river systems in eastern Canada originate as small, remote headwater systems, and these habitats are important to recruitment and production of fish populations. These types of habitats, however, have received little attention in the study of interactions between forest harvesting and the biotic communities. A multi-disciplinary research project is currently under way in a small headwater ecosystem in Newfoundland, Canada (Copper Lake Watershed) to address the ability to ameliorate the effects of forest harvesting through the provision of an unharvested riparian leave strip, or buffer zone. A major component of this study is the examination of sediment delivery associated with road construction and harvesting operations. This paper outlines initial changes observed after road construction and limited clear cutting within the Copper Lake watershed.

Methods

The present study was undertaken as part of a large multi-agency, multi-disciplinary research project in western Newfoundland known as the *Copper Lake Buffer Zone Study*. The Copper Lake watershed is a small headwater system (13.5 km^2) located about 17 km southeast of Corner Brook, Newfoundland, Canada (Fig. 1). Details on the study area, as well as a description of the Copper Lake Buffer Zone Study, are provided in Scruton et al. (1995).

Fine particulate sediment accumulation was monitored using the method described by Wesche et al. (1989) with sediment samplers consisting of modified Whitlock-Vibert boxes (14 × 6.4 × 8.9 cm deep, with 3.5 × 13 mm openings). These boxes (typically used for egg incubation) were filled with cleaned gravel (about 25 mm in diameter). A strip of duct tape on the bottom of the boxes prevented loss of accumulated fines. Sediment traps were retrieved by lifting them to the surface, being careful not to lose the accumulated fine sediment, and immediately placing them in plastic collection bags for later analysis. Samples were wet sieved through a set of standard nested sieves (2.50, 1.40, 0.85, 0.50, and 0.09 mm), dried at 70°C for 24 hours, and weighed in each of the four size fractions (<1.40, <0.85, <0.50, and <0.09 mm diameter).

In the first year of the study (1993) 15 sediment boxes were deployed in each of three stream reaches: upstream of the proposed road crossing on T1 (T1U), downstream of the road crossing (T1L), and in a stream reach (T1-1L) not affected by the road in 1993 (Fig. 1). Sediment boxes were arranged in five stations longitudinally distributed in each stream reach. There were three boxes per station distributed across the wetted width of the channel. Sediment boxes were collected, then replaced in October 1993 after road construction for that year was complete. Sediment traps were re-deployed (October 1993) in T1L and T1-1L while monitoring of T1U was discontinued. At the same time sediment traps were deployed in T1-1U, T1-3L, and T1-3U (Fig. 1) to expand sedimentation monitoring as part of the larger buffer zone study (Scruton et al. 1995).

Continuation of road construction in 1994 allowed for replication of the perturbation experiment carried out in 1993 as the road bisected a second tributary (T1-1) in the spring of 1994. Sediment

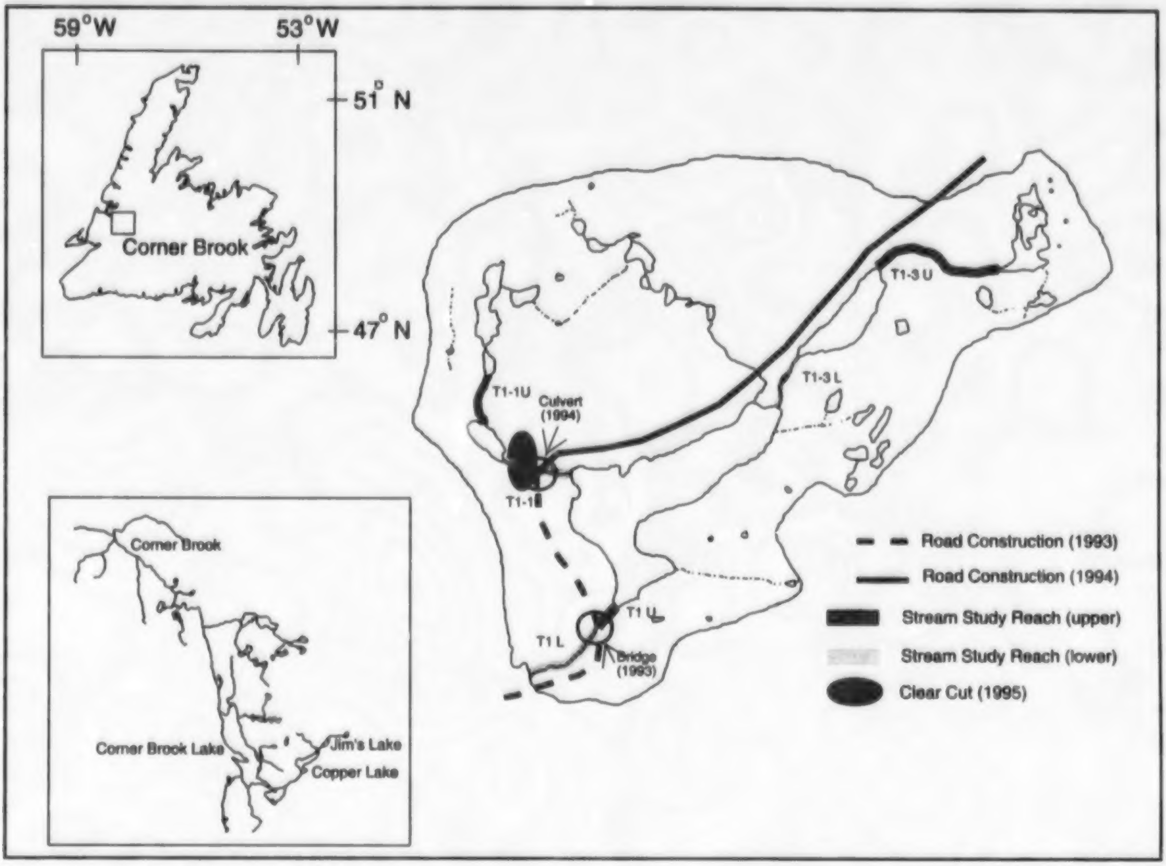

Figure 1. Location of the Copper Lake watershed with the road construction and areas of forest harvest highlighted.

samplers were collected, then re-deployed in both June and October 1994, thus encompassing two seasonal periods: the open water, low flow period from June to October (October sample) and the overwinter (ice covered) period, including peak spring and fall flows (June sample). This sampling regime allowed for seasonal comparison of sediment accumulation in these streams while at the same time allowing for continual monitoring of the effects of road construction on T1L and T1-1L.

The third year of the study (1995) allowed for a continuation of monitoring of sediment increases due to road use on T1 and base line data for T1-1U, T1-3L, and T1-3U in correspondence with the overall study design (Scruton et al. 1995). In addition, the top 100 metres of T1-1L or 20% of its length (above the road crossing) was cut to the water's edge in the fall of 1994 after the October sample was collected. This allowed the evaluation of complete forest harvesting on sediment accumulation in this small stream. The sampling protocol was the same as that established in the previous years, with sediment samplers being collected and redeployed in June and October 1995.

Visual inspection of the data distributions collected from the Whitlock-Vibert boxes revealed that the data were not normally distributed in all cases. To avoid assumptions of distribution that might not have been met, randomization with replication was used to estimate 95% confidence intervals of the mean for each stream reach (Sokal and Rohlf 1981; Edgington 1987). The raw sediment accumulation (g) for each box collected in a particular stream reach was replicated 200 times and produced a population of 1600–3000 numbers depending on the number of traps retrieved (N), which ranged from 8 to 15. From these populations, N numbers were randomly sampled and their mean calculated 1000 times. Statistical

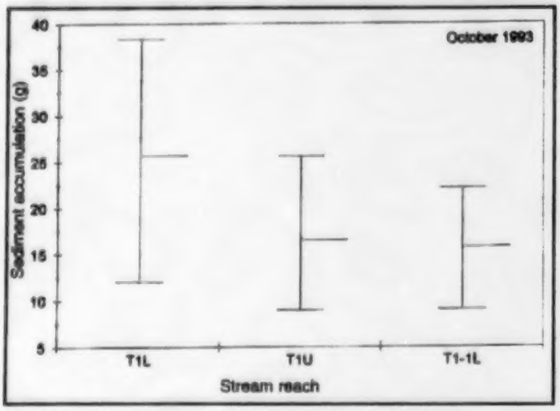

Figure 2. Sediment accumulation in the experimental streams in October 1993. Boxes represent 95% confidence intervals around the mean.

significance between stream reaches was determined graphically, where a failure of the mean of one distribution to overlap the confidence interval of another distribution was considered significantly different at $p = 0.05$.

Benthic macroinvertebrate abundance was evaluated by deploying artificial substrates in five locations on each of T1, T1-1L, T1-3L, and T1-3U in May 1994 (Scruton et al. 1995). Each substrate consists of a plastic tray filled with cobble-sized substrate from the adjacent stream bed (Ryan et al. 1985). The artificial substrates were harvested in October 1994 and 1995, specimens were sorted, counted, and identified to genus where possible using keys in Merritt and Cummins (1984). Taxonomic lists were developed for each study stream and the average abundance for the major orders was compared between stream reaches. The confidence intervals for average abundance were estimated using randomization with replacement techniques.

Brook trout population estimates were determined by electrofishing using the fixed effort (successive) removal method. Barrier nets cordoned off each station to prevent migration to/from the study site. Successive sweeps (runs) at each site were made using a backpack electrofisher, with a minimum of four sweeps per site. Electrofishing equipment (Smith-Root Type 12 model) and methods are described in detail in Scruton and Gibson (1995). Population estimates were calculated using the Microfish 3.0 program developed by the U.S. Fish and Wildlife Service (Van Deventer and Platts 1989), which employs a maximum likelihood (ML) estimator (Burnham formula, Van Deventer and Platts 1983).

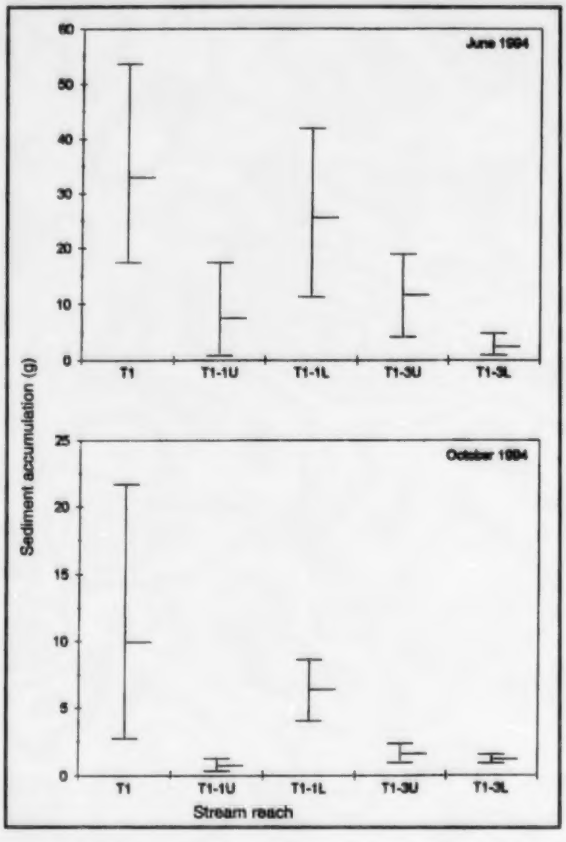

Figure 3. Sediment accumulation in the experimental streams during 1994. Boxes represent 95% confidence intervals around the mean.

Results and Discussion

Sediment Accumulation

The total sediment accumulation below the newly constructed arch culvert on T1 (T1L) in October 1993 was 1.6 times the accumulation above the road crossing (T1U) (Fig. 2). This difference was statistically significant ($p < 0.05$). The T1U and the unaffected stream T1-1L had similar total sediment yields (Fig. 2).

Elevated sediment accumulation was observed in tributary T1-1L in June 1994 after a culvert was installed in the stream (Fig. 3). Tributary T1L still had an elevated sediment accumulation in June 1994 but not significantly different from that of T1-1L. The undisturbed stream reaches T1-1U and T1-3U had similar sediment yields in June 1994 while T1-3L had a significantly lower sediment accumulation

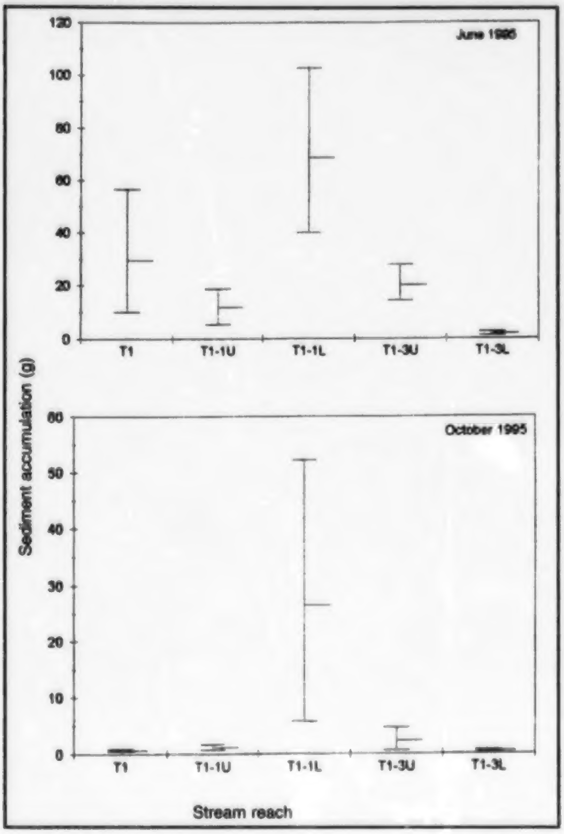

Figure 4. Sediment accumulation in the experimental streams during 1995. Boxes represent 95% confidence intervals around the mean.

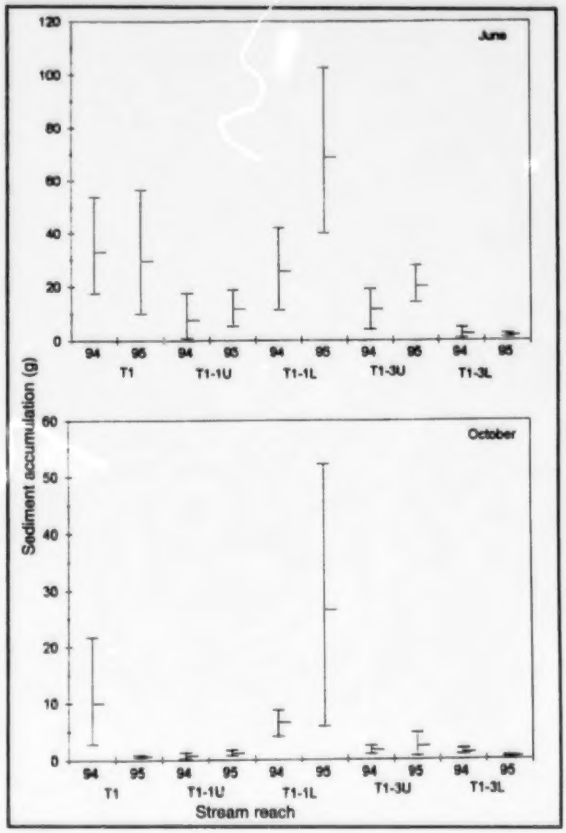

Figure 5. Inter-annual sediment accumulation, comparing the same sampling dates in 1994 and 1995.

than the other two (Fig. 3). The October 1994 sample revealed a similar pattern to that observed in June 1994, with T1 and T1-1L having significantly higher total sediment accumulations than the other stream reaches (Fig. 3). Thus the initial effect on sediment accumulation with the two types of construction techniques appeared to be similar.

In June 1995, after limited forest harvesting, the sediment deposited in T1-1L was significantly higher than in all the other streams. The accumulation observed in T1L was still elevated compared to the other streams but was not as high as the accumulation observed in T1-1L (Fig. 4). The sediment accumulation had returned to normal levels in T1L by October 1995 while the accumulation in T1-1L was still significantly higher than the accumulation in all the other streams reaches (Fig. 4). The road crossing and limited clear cut have confounded these results and might have combined to increase sediment accumulation in T1-1L as compared to T1L and the other streams.

Sediment accumulation was greater after the spring freshet (June) than was observed for the summer low-flow period October (Figs. 3 and 4). Seasonal differences in sediment accumulation for the undisturbed stream reaches T1-1U, T1-3L, and T1-3U were 10.2, 2.1 and 7.2, respectively in 1994. The smallest decrease from the June to October sample was observed in T1-3L (2.1), which was due to consistently lower sediment accumulation throughout the monitoring period. A similar pattern was observed in 1995 although the difference observed in T1L was much higher than was observed in 1994 (Fig. 4) at the road crossings T1L and T1-1L. Seasonal patterns of sediment accumulation at these sites were similar to the other stream reaches, but the magnitude of decrease from June to October was

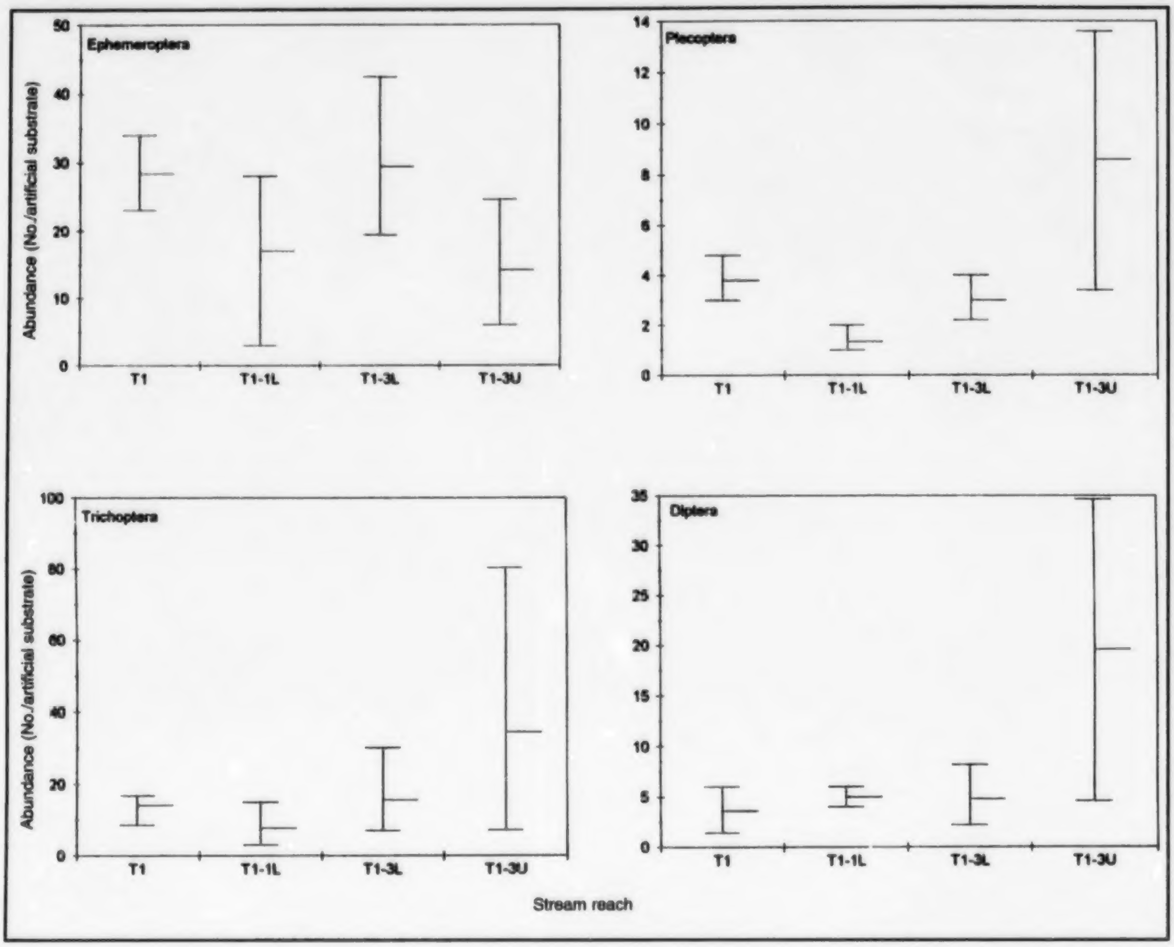

Figure 6. Abundance (No./artificial substrate) of the most common benthic macroinvertebrates in the stream reaches during the summer of 1994. Boxes represent 95% confidence intervals around the mean.

lower, 3.3 and 4.0, respectively in 1994 (Fig. 3) and 2.6 in 1995 for T1-1L (Fig. 4). Thus, the increased sediment accumulation over the summer may be of more importance to the biological populations in these habitats because it does not occur in the natural environment, while increased sediment after spring runoff is a natural occurrence. Also, this is the most important time for growth and development for biological populations using these habitats.

There was very little inter-annual variation observed in sediment accumulation between corresponding sampling dates in the unperturbed streams (Fig. 5). This observation was in spite of a large hydrological event in early June of 1995. In the perturbed streams no difference was observed in the June samples for T1 while the accumulation of sediment in T1-1L was significantly higher in 1995, probably due to the limited clear cutting on that stream reach (Fig. 5). When comparing the October samples we observed a significant reduction in sediment yield in T1 and a significant increase in T1-1L from 1994 to 1995 (Fig. 5). This suggests that the limited clear cut has increased the sediment accumulation on T1-1L, and the total accumulation for this stream is due to a combination of the road crossing and the clear cut.

Initial Analysis of Sediment and the Biotic Communities

The abundance of the major macroinvertebrate taxa did not show any clear trend related to

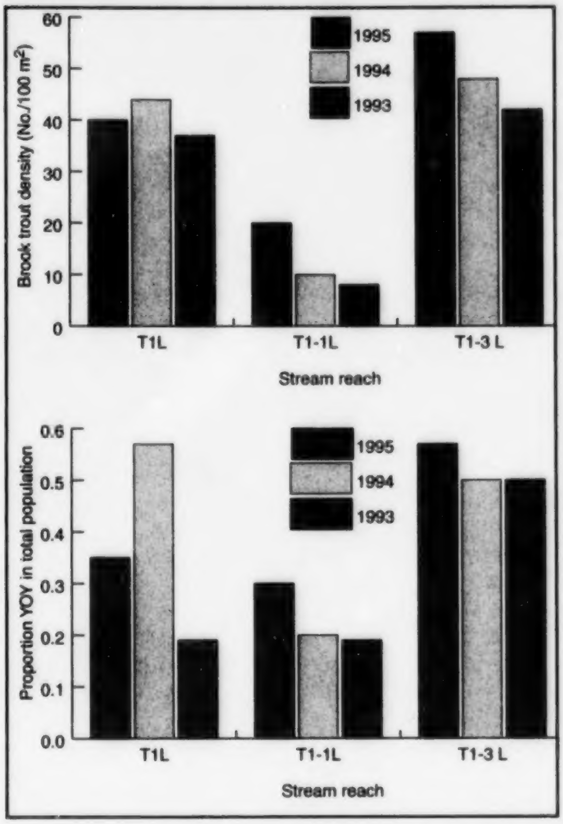

Figure 7. Brook trout population density (top) and the proportion of young-of-the-year trout in the population (bottom) in the two perturbed streams (T1L and T1-1L) and a control (T1-3L).

increased sediment accumulation in T1L and T1-1L during 1994 (Fig. 6). T1-1L did have significantly fewer Plecoptera and Trichoptera than did the other streams reaches but these observations are not conclusive because there were significant natural differences in benthic abundance between the control streams. The possibility of reduced macroinvertebrate abundance due to increased sediment accumulation in these streams will be closely monitored in the future.

Brook trout densities in T1-1L continuously declined over the course of this study while those in T1L have been variable (Fig. 7). The trout densities of T1-3L, a control stream with a continually low sediment yield, have also been observed to be declining (Fig. 7), and thus there was no strong relationship between increased sediment accumulation and total brook trout density over the entire stream reach in the initial stages of forest harvesting. However, the increased sediment reduced the proportion of young-of-the-year (YOY) fish in T1-1L compared to the proportion in T1L and T1-3L (Fig. 7). This effect appeared to be most pronounced closer to the sediment source (i.e., the road crossing). The 100-metre section of stream just below the road crossing has had severe reduction in brook trout densities, from 19.63 in 1993 to 7.53 in 1994 and 2.40 per 100 m^2 in 1995, and there have been no young-of-the-year trout caught in this section since the road construction. This may be due to increased movement of older fish to downstream and lake sites in this tributary (McCarthy et al. 1998) and because the young-of-the-year are restricted from this section by the high sediment accumulations. These observations in the area adjacent to the perturbation may give us some insight into the results we can expect once the rest of the stream reach is clear cut.

Conclusions

Logging road construction significantly increased the sediment accumulation in the small headwater streams of the Copper Lake watershed. These increases were observed at two independent stream crossings, one using an arch culvert and the other a whole cylindrical culvert. The two types of stream crossings had similar effects on sediment accumulation immediately after road construction but the stream with the arch culvert (T1L) appears to be returning to predisturbance levels after two years. A limited clear cut of 20% of its length above the road on T1-1L in 1995 increased sediment accumulation to an even higher degree than was observed in 1994. This development will confound any future comparisons between the two crossing types.

Sediment accumulation was significantly higher in the June samples compared to the October samples, but there was very little inter-annual variation between years in the uncut streams. Road construction and the limited clear cut reduced the natural seasonal difference in sediment accumulation. Differences observed in the October samples in the affected streams were more pronounced than those observed after the spring freshet (June samples). The pronounced October differences occurred despite a large hydrological event in early June of 1995 and suggests that the increased sediment over the summer in the perturbed streams may be of more importance to the biological populations.

There were no discernible trends in benthic macroinvertebrate abundance and increased

sediment accumulation during 1994. There were lower levels of Plecoptera in T1-1L, but this was not conclusive and requires further investigation. Brook trout densities have, however, been reduced in T1-1L and the young-of-the-year trout and habitats in close proximity to the sediment source are at the highest risk.

Acknowledgments

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A Summary of the Effects of Forest Harvesting, Fishing, and Environmental Shifts on Salmonid Populations of Carnation Creek, Vancouver Island, British Columbia

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Abstract

The Carnation Creek Fisheries-Forestry Interaction Project was initiated in 1970 and is the longest, continuous study of the effects of forestry practices on biological and physical watershed processes in North America. This case study was initially designed to investigate the effects of different streamside forest-harvest treatments on stream channels, aquatic habitats, and fish. One component of this multi-disciplinary study has been the monitoring of the salmonid populations of Carnation Creek through 5 pre-logging, 6 during-logging, and 14 post-logging years. Forest harvesting has had complex and often variable effects upon Carnation Creek fish species and life stages. Chum salmon (*Oncorhynchus keta*) have shown the sharpest decline. Numbers of adults returning to the stream after logging fell to about one-third of the pre-logging average. This decline is due partly to reductions in egg-to-fry survival resulting from decreased quality of spawning and egg-incubation habitats in the lowermost stream reach. Reductions in summer rearing habitat appear to explain the roughly 50% post-logging decline in abundance of coho salmon fry (*O. kisutch*) inhabiting the stream. However, fewer coho fry have produced >1.5- times more smolts after logging due to improved overwinter survival, which is in turn correlated with increased winter water temperatures and summer growth. Increased smolt production has not caused more adults to return. Coho salmon returning to the system have declined after logging by 31%, due partly to depressed marine survivals resulting from earlier timing of spring smolt migrations and ocean climate shifts. The production of salmonids from coastal streams clearly depends upon processes occurring both within watersheds and the marine environment. We cannot control natural shifts in marine ecosystems and climate. Therefore, to maintain our salmonid resources, we must always apply our best forest-harvest practices to ensure that adverse effects of natural variations are not compounded with those of inappropriate land use.

Tschaplinski, P. J. 1998. A summary of the effects of forest harvesting, fishing, and environmental shifts on salmonid populations of Carnation Creek, Vancouver Island, British Columbia. Pages 361–388 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

The effects of forest harvesting on fish populations have been studied for over 25 years at Carnation Creek on the west coast of Vancouver Island, British Columbia. This single-watershed, intensive case study has generated the longest series of continuous data on fisheries–forestry interactions anywhere. The Carnation Creek Experimental Watershed Project was initiated in 1970 by the federal agency now known as Fisheries and Oceans Canada. The project soon expanded into a multi-agency, multi-disciplinary program on the effects of forest harvesting on a coastal watershed and its salmon and trout populations. In the 1960s, resource managers and planners based judgments on the effects of logging on fish populations from studies conducted elsewhere in North America; for example, in Oregon, Alaska, and as far away as New Hampshire. Both the forest industry and government resource agencies expressed concern that these extrapolations might not lead to the most appropriate planning decisions for areas on the west coast of British Columbia. Therefore, the Carnation Creek study was initiated in order to provide fisheries–forestry information on at least one type of drainage basin in coastal B.C.

The objectives of the Carnation Creek study were to: (1) provide an understanding of the physical and biological processes operating within a coastal watershed; (2) reveal how the forest harvesting practices employed in the 1970s and early 1980s changed these processes; and (3) apply the results of the study to make reasonable and useful decisions concerning land-use management, fish populations, and aquatic-habitat protection. The project has achieved these goals despite limitations associated with intensive studies made only in a single watershed. Over 180 publications have been produced from Carnation Creek research. The results from this project have made major contributions to the British Columbia Coastal Fisheries–Forestry Guidelines (CFFG) implemented in 1987, and the legally binding provisions for aquatic habitat protection within the new British Columbia Forest Practices Code implemented in 1995, which have replaced the CFFG.

Fish populations have been studied at Carnation Creek on a virtually continuous basis since 1970. The objectives of this review are to illustrate: (1) the changes in the abundance, growth, and survival of coho (*Oncorhynchus kisutch*) and chum salmon (*O. keta*) in Carnation Creek between 1970 and 1995 through 5 pre-logging, 6 during-logging, and 14 post-logging years; (2) that the effects of forest harvesting are complex, and vary among life stages and species; and (3) that salmonid production depends on biological and physical processes occurring both within watersheds and in marine environments (e.g., climate-associated changes, predation, and fishing). Long-term trends in the abundance of anadromous rainbow (steelhead) trout (*Oncorhynchus mykiss*) and cutthroat trout[1] (*O. clarki*) are also discussed briefly. The following discussion will also demonstrate the value of long-term, multi-disciplinary studies for clarifying complex interactions among land-use practices and natural processes occurring within watersheds, which determine salmonid abundance and growth in coastal streams.

Methods

The design of the Carnation Creek Experimental Watershed Project and the methods employed for monitoring physical variables and biological processes before, during, and after forest harvesting are thoroughly described by Hartman and Scrivener (1990). Hartman and Scrivener (1990) summarized all aspects of the project and provided a comprehensive bibliography of the publications generated from Carnation Creek research up to 1990. The following summary is condensed from the detailed descriptions of the project published by Hartman and Scrivener (1990).

Carnation Creek is located approximately 20 km northwest of Bamfield on the south shore of Barkley Sound in southwestern Vancouver Island (Fig. 1; 49°N, 125°W). The watershed occurs within the Coastal Western Hemlock Biogeoclimatic Zone, which spans the west coast of North America from the Olympic Peninsula in Washington State to the Queen Charlotte Islands and southeast Alaska (Krajina 1969). The stream drains an area of 11 km^2 and contains rugged terrain between 0 and 800 m in elevation. The valley walls have gradients up to 80%. The main stream is approximately 7.8 km long, but only the lowermost 3.1 km, extending from the stream mouth to the base of a steep-gradient canyon, is inhabited by anadromous salmonids including coho salmon, chum salmon, rainbow (steelhead) trout, and cutthroat trout. This lowest stream reach contains a valley bottom of about 55 ha that is 50–200 m wide. The coarse,

[1] For brevity, coho salmon, chum salmon, rainbow (or steelhead) trout, and cutthroat trout are frequentlyt referred to in this paper as coho, chum, rainbow, steelhead, and cutthroat, respectively.

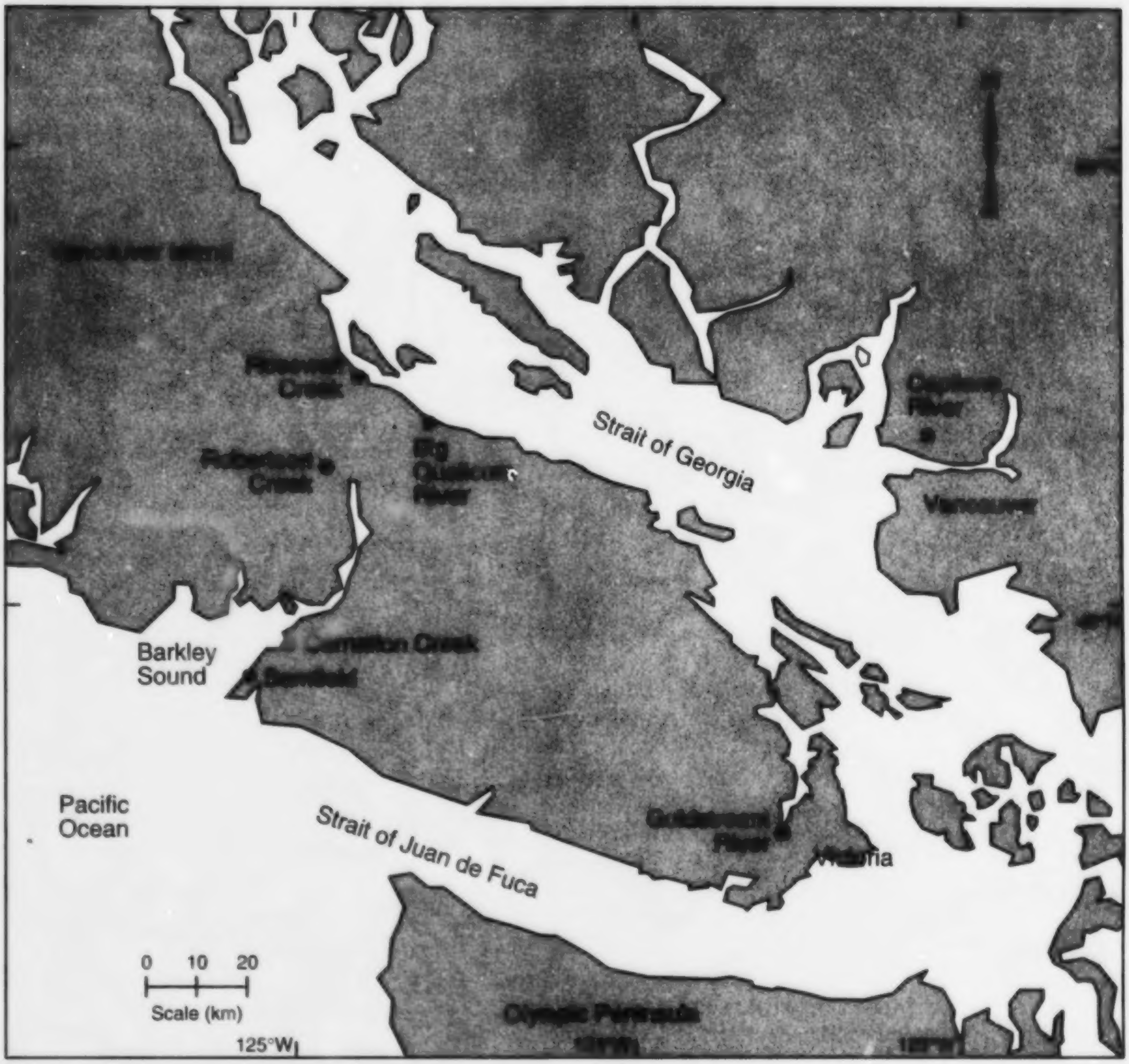

Figure 1. Location of the Carnation Creek study site in southwestern Vancouver Island, B.C.

well-drained soils, forest cover, hydrology, and heavy annual precipitation (varying from 210 to over 500 cm/yr) are typical of western Vancouver Island and many other areas of coastal B.C. About 95% of the annual precipitation falls as rain, primarily during autumn and winter. High variations in seasonal rainfall cause stream discharge to range from 0.03 m^3/s in summer to 64 m^3/s in winter. Stream flows may increase 200-fold within 48 h because of rapid runoff of rain from storms that can produce up to 26 cm of precipitation within the same time period.

The Carnation Creek study was designed initially to examine the effects of three different types of streamside forest-harvest treatments on stream channels and fish populations. These treatments were applied along the lowermost 3 km of the stream (Hartman and Scrivener 1990). A leave-strip treatment was applied from the estuary to 1 300 m upstream. This treatment was designed to buffer the effects of clearcut logging from the stream channel by leaving a riparian strip of trees that varied from 1 to 70 m wide. An intensive treatment was applied along 900 m of stream channel immediately upstream from

the leave-strip treatment. In the intensive treatment, clearcut harvesting occurred simultaneously along both sides of the stream up to the channel margin. No riparian trees were left standing. Any activity within the channel that was considered operationally convenient, such as felling and yarding trees across the stream, was permitted. Economically valuable, wind-thrown trees lying within the stream channel were removed. Logging-associated debris was burned after harvesting was completed. The third treatment, called careful clearcutting, was applied over the 900 m length of stream immediately upstream from the intensively treated area. No activity within the stream was permitted in this treatment with the exception of the felling and removal of six trees leaning over the stream channel. Perennial vegetation on the streambanks, such as salmonberry (*Rubus spectabilis*), was left alone; however, red alder trees (*Alnus rubra*) were removed from the streamside.

The responses of a comprehensive set of biological and physical variables within the Carnation Creek basin were determined relative to forest harvesting over (1) 5–6 pre-logging years spanning 1970–1975 (beginning in 1970 or 1971, depending upon the variable measured), (2) 6 years spanning 1976–1981 during which 41% of the watershed was harvested (including almost all of the valley bottom), and (3) 14 post-logging years from 1982 to 1995. From 1987–1995, an additional 21% of the basin was harvested. This later harvesting occurred in headwater areas remote from the main stream channel.

Historical Data Collections

Data collected historically have included comprehensive information on climate; stream temperatures and discharge; groundwater levels (by using piezometers); water chemistry; stream channel morphology and large woody debris abundance and distribution; streambed particle-size composition (by using frozen-core methods); suspended sediment transport during high flows (by using automated sampling at one hydrological weir); streambed scour and deposition; ground disturbance, landslides, and post-logging revegetation; biomass of aquatic algae (periphyton); abundance and distribution of benthic macroinvertebrates; and fish populations. Details of all historic methods are given in Hartman and Scrivener (1990).

Fish population studies have included: (1) abundance and distribution of adult salmonid spawners returning to the stream (autumn and winter); (2) numbers of juvenile fish [salmonid smolts and young-of-the-year (fry)] migrating seaward in spring; (3) abundance, distribution, age structure, growth, and survival of juvenile salmonids rearing in freshwater and estuarine habitats during summer and early autumn; (4) seasonal movements of juvenile salmonids out of the main stream into off-channel overwinter habitats, and return movements in spring; (5) main-channel and off-channel abundance, distribution, and survival of juvenile salmonids in winter; (6) chum egg incubation, egg survival, and fry emergence (redds capped with trap nets); and (7) fecundity determinations for female chum and coho salmon for estimates of annual egg-to-fry survival (Andersen 1978, 1981, 1983, 1987; Andersen and Narver 1975; Andersen and Scrivener 1992; Brown 1987; Brown and Hartman 1988; Brown and McMahon 1988; Bustard 1991; Bustard and Narver 1975; Tschaplinski 1982a, 1982b, 1988; Tschaplinski and Hartman 1983).

Current Data Collections

Many variables and processes continue to be studied at this research site. Since 1990, data have collected on fish populations and habitat, stream channel morphology, streambed movements (plus erosion and sedimentation), climate, hydrology, forest regeneration and growth, and hillslope processes.

Water temperature, depth, and discharge were monitored at permanent hydrological weirs installed on the mainstream and on several principal tributaries. Climate stations are located in several sites at different elevations in the watershed. Some stations are co-located with the hydrological weirs. Climate variables monitored include air temperature, solar radiation, precipitation, relative humidity, and wind speed and direction. Climate and hydrology stations have been updated by the installation of continuous-operation, electronic data recorders (Tschaplinski et al. 1996).

Changes in channel morphology were determined annually in nine survey reaches of the stream (which incorporate the same sections used to determine seasonal fish population abundance and distribution). Standard survey-and-mapping techniques were employed (see Scrivener and Hartman 1990). Within each survey section (1) all pieces of large woody debris (LWD, which includes tree trunks, root masses, and large limbs) were mapped and tagged in order to observe changes in distribution and abundance, and (2) textural distributions of surface sediments were described visually by using grid samplers (Hogan 1996). Each year, these ground-based surveys were supplemented with aerial photographic surveys of the entire creek channel. Stereo photographs

(70-mm aperture) were used to determine changes in channel structure in areas between study reaches and to generate inventories of fish habitats throughout the stream. Aerial photographs are also being employed to monitor the rates of canopy closure over the creek as the new forest grows. Canopy closure and forest growth will be related to future water temperature changes in Carnation Creek.

Scour-and-fill monitors were used to study channel scour and deposition in the same survey reaches (Haschenburger 1996). Estimates of sediment (bedload) transport were determined by following the annual movements (distances and depths) of painted, magnetic rocks placed onto the streambed and which represent the range in sizes and textural proportions within each study reach (Haschenburger 1996).

Adult salmonids (coho and chum salmon, and steelhead and cutthroat trout) returning to spawn in Carnation Creek were enumerated accurately at the main fish weir (fence) located near the mouth of the stream at the tidewater limit. The species and sex of the spawners were identified. Ages were determined from scale samples, and lengths were taken. Every day, observers visually enumerated chum salmon spawning downstream of the fence (Tschaplinski et al. 1996).

Juvenile salmonids (fry and smolts) and sculpins (*Cottus asper* and *C. aleuticus*) migrating seaward in spring were also enumerated and their species identified at the main fish weir. Large samples of salmonid fry and smolts (up to 50 individuals of each species per day) were measured for length and weighed. Scale samples of up to 50 smolts of each species were taken daily.

The abundance, habitat distribution, growth, and survival of juvenile salmonids and sculpins rearing in Carnation Creek from spring to autumn were determined from two to three seasonal surveys usually conducted between 15 June and 30 September. During each survey, the two-catch removal method (Seber and LeCren 1967) was used to assess abundance within nine to ten representative study sections within Carnation Creek (95% confidence limits usually within 5–10% of the estimate). Fish were captured by electrofishing and seining in each of two fishing trials, identified as to species, and measured for length (see Andersen and Narver 1975; Tschaplinski et al. 1996). Large samples were weighed, and scales were taken to determine population age–size distributions and age-specific growth rates. The total abundance of fish in Carnation Creek was determined by extending the numbers of fish captured in the survey sections to the total length of stream inhabited by each species. Within each surveyed section, the total wetted surface area of the stream and its component pool, glide, and riffle areas were measured to determine population densities for specific habitat units. Fish habitat was classified and quantified according to methods adapted from Bisson et al. (1982) and Hankin and Reeves (1988).

Overwinter survival of juvenile salmonids was determined from the difference between population abundances estimated in late summer and the numbers of smolts migrating seaward from Carnation Creek in spring (plus any residual parr remaining in the stream in spring which were estimated by the population surveys). Seasonal changes in distribution and habitat use between summer (rearing) and winter (shelter) were determined each year by juvenile population surveys in the main channel in winter, and by monitoring the movements of salmon and trout between Carnation Creek and its valley-bottom tributaries by daily enumerations of fish at tributary weirs. Winter abundances of fish in specific off-channel sites were determined by employing removal methods and large numbers of Gee-type fish traps baited with salted fish roe. The seasonal use of these off-channel sites by fish, and changes in habitat characteristics were determined annually.

Results and Discussion

The principal trends in Carnation Creek fish population abundance, distribution, and survival over the past 25 years are discussed primarily for coho and chum salmon, which are the numerically dominant species in the watershed.

Adult Chum and Coho Salmon Returns

Adult chum return to spawn in Carnation Creek primarily at age 4 (4-year-olds, usually > 80%), and mainly in October and November (Hartman and Scrivener 1990). In most years, chum salmon have been the numerically dominant salmonid spawning in Carnation Creek (Fig. 2). However, this species has shown the most drastic decline in abundance after forest harvesting. Prior to forest harvesting (Fig. 2; 1970–1975), adult chum returns ranged from 1 000 to 4 168 and averaged 2 188 (95% confidence interval: ±1 272)[2]. During the 6 years of logging (1976–1981), chum returns were not significantly different from the pre-logging average: mean numbers

[2] Variation associated with mean values are ± 95% confidence limits in this paper unless otherwise noted.

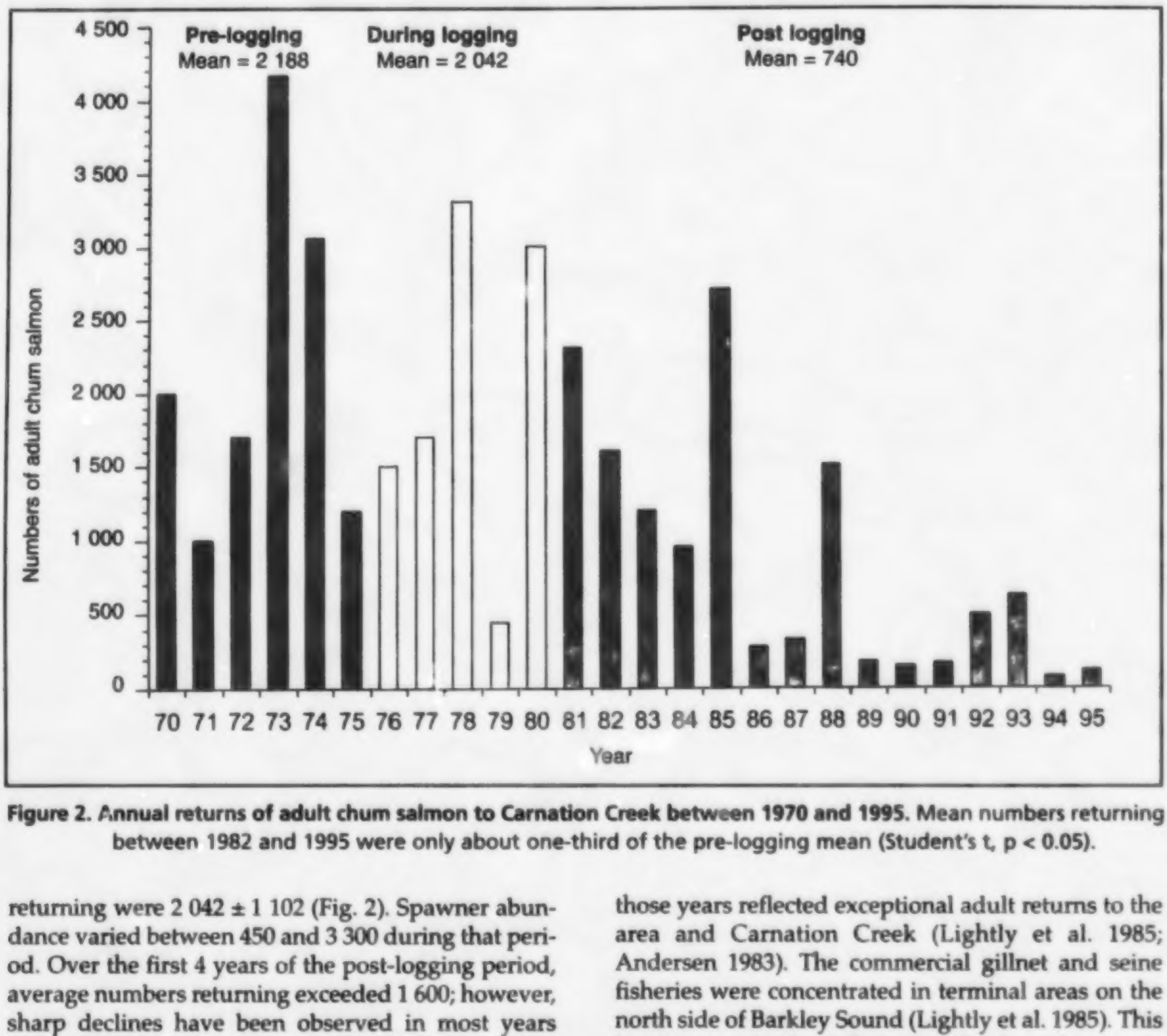

Figure 2. Annual returns of adult chum salmon to Carnation Creek between 1970 and 1995. Mean numbers returning between 1982 and 1995 were only about one-third of the pre-logging mean (Student's t, $p < 0.05$).

returning were 2 042 ± 1 102 (Fig. 2). Spawner abundance varied between 450 and 3 300 during that period. Over the first 4 years of the post-logging period, average numbers returning exceeded 1 600; however, sharp declines have been observed in most years since 1986. On average, only 740 ± 444 chum have returned to spawn in Carnation Creek in the post-logging period (1982–1995). Therefore, chum returns have averaged only about one-third of their pre-logging levels (Student's t, $p < 0.05$), and were less than one-sixth of the pre-logging average in 6 of the 14 years for which post-logging data are available.

Fisheries on Barkley Sound chum salmon are thought to have had little effect on the numbers of this species returning to Carnation Creek. Commercial harvesting in Barkley Sound has been restricted since 1962 (Lightly et al. 1985). Lightly et al. (1985) reported that the fishing rate has been <0.01 (<1%) in 15 of 24 years examined, and usually <15 % in most years since 1951. In 1971, 1973, 1978, and 1980, fishing took an estimated 20–43% of Barkley Sound chum salmon, but the extensive fishery in those years reflected exceptional adult returns to the area and Carnation Creek (Lightly et al. 1985; Andersen 1983). The commercial gillnet and seine fisheries were concentrated in terminal areas on the north side of Barkley Sound (Lightly et al. 1985). This location was far from Carnation Creek, and suggests that few chum salmon from Carnation Creek were caught.

Annually, local aboriginal peoples have conducted a small food fishery for chum; however, this fishery often consisted of only one net-set during the peak of the adult run returning to the nearby Sarita River (1.8 km away). Up to 300 fish are taken annually, and some are probably Carnation Creek chum salmon (Holtby and Scrivener 1989). However, the long-term decline in chum salmon returns to Carnation Creek coinciding with the post-logging period is not likely due to fishing mortality.

Two patterns in coho spawner abundance have been observed, and each is associated with one of two types of coho returning annually to Carnation Creek. In most years, the majority of coho spawners

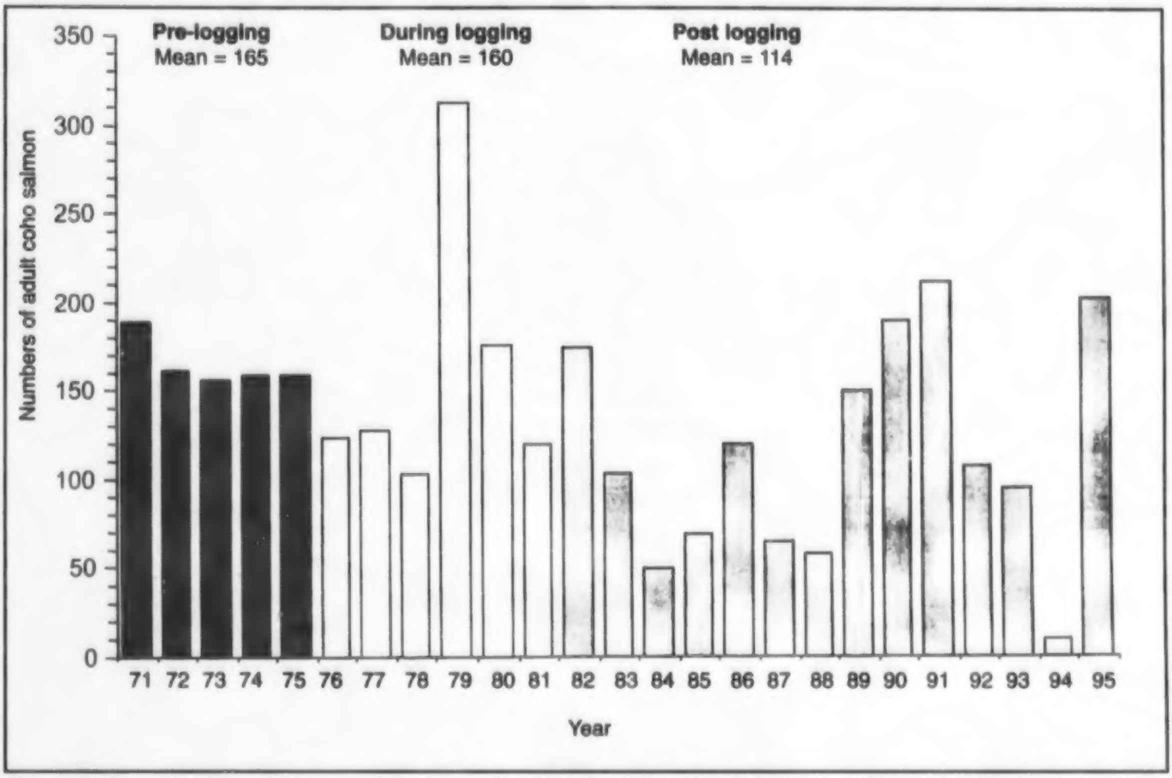

Figure 3. Annual returns of large adult coho salmon to Carnation Creek between 1971 and 1995. Mean numbers returning during the post-logging years were significantly lower (by about 31%) than returns in the pre-logging period (Student's t, $p < 0.05$).

are age-3 or age-4 adults that have spent roughly 18 months, including two summers, in the ocean (Hartman and Scrivener 1990). These fish are called large adult coho because most individuals are > 44 cm long (fork length). Other coho, usually ≤ 44 cm long, return to spawn after spending only about 5 to 6 months in the ocean (Hartman and Scrivener 1990). Most of these small fish are 2-year-old precocious males called jacks.

Returns of large adult coho have declined significantly after logging in Carnation Creek (Fig. 3; Student's t, $p < 0.05$); however, the decline has been less marked than that shown by chum. Before forest harvesting (1971–1975), 165 ± 17 large adult coho returned on average each year (Fig. 3). These returns decreased by about 31% to only 114 ± 36 in the post-logging period. Most of the decline is due to decreased returns of females. In pre-logging, during-logging, and post-logging periods, the mean numbers of females returning to spawn were 73 ± 6, 74 ± 53, and 48 ± 16, respectively. The abundance of adult females has declined on average by about 34% in the post-logging period compared with numbers in pre-logging years (Student's t, $p < 0.05$). In the pre-logging and during-logging periods, the male:female ratios were 1.25:1 and 1.20:1 on average. The proportion of males increased in the post-logging period to 1.42:1 on average[3] and varied from 1:1.47 (i.e., females exceeding males) to 2.2:1. The long-term decline in females has not been explained; however, it appears to be widespread among coho stocks from streams in the Barkley Sound area and to the south (Simpson et al. 1996).

In contrast with large adults, the mean numbers of jacks returning show no statistically significant trend among pre-logging, during-logging, and post-logging periods (Fig. 4; Student's t, all $p>0.05$). The numbers of jacks actually exceeded the numbers of large adult coho in 1978, 1988, 1989, and 1994 when they made up respectively 69.6, 63.2, 50.5, and 92.0%

[3] Excluding the unusually low returns of 1994 when only 1 female and 8 adult males were observed in the stream.

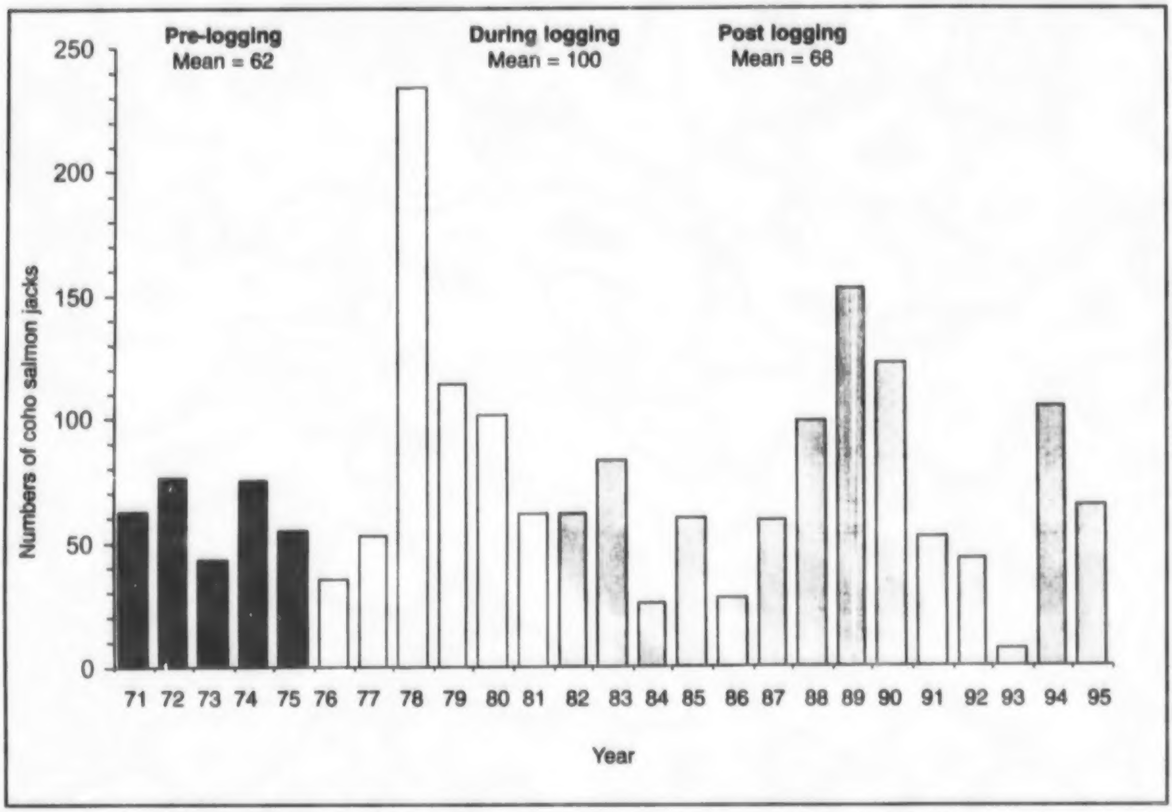

Figure 4. Annual returns of coho salmon jacks to Carnation Creek between 1971 and 1995. Mean numbers returning were not significantly different among study periods (Student's t, all $p > 0.05$).

of the total numbers of coho returning to spawn. Therefore, when large adults and jacks are combined, the significant decline in the total coho return to Carnation Creek between pre-logging (227 ± 26) and post-logging (182 ± 45) periods is due to the decline in numbers of large adult coho alone (Fig. 5; Student's t, $p < 0.05$). On several occasions, strong returns of jacks in a given year have been followed by strong returns of large adults in the next year (compare brood years for 1979 jacks with 1980 adults, and 1994 jacks with 1995 adults; Figs. 3 and 4); however, this pattern was not consistent among all years.

The abundance of large adult coho was nearly invariant during the years prior to logging (Fig. 3). However, the interannual variation in the abundance of both jacks and large adults has increased dramatically since 1976 when forest harvesting activities were initiated (Figs. 3, 4). The increase in variation in the during-logging and post-logging periods included especially low adult returns observed in 4 of 5 years spanning 1984 and 1988 when large coho averaged only 60 at Carnation Creek (Fig. 3). These depressed numbers occurred at roughly the same time during the mid-1980s when chum salmon returns also began to show sharp declines in some years, although species-specific differences in annual patterns were observed (Figs. 2 and 3). In contrast with chum, coho spawner abundance increased dramatically in the next 3 years (1989–1991) during which time the number of adults returning exceeded the pre-logging average in each year (Fig. 3). However, these elevated returns completely reversed in the following 3 years when Carnation Creek coho (and chum) were subjected to the simultaneous effects of forest-harvesting, fishing, and prolonged poor conditions for marine survival caused by the northward extension of warm, nutrient-poor waters from southern latitudes (Hargreaves and Hungar 1994; Rice et al. 1995). The warm-water phenomenon known as El Niño was characterized not only by low ocean productivity but also by elevated

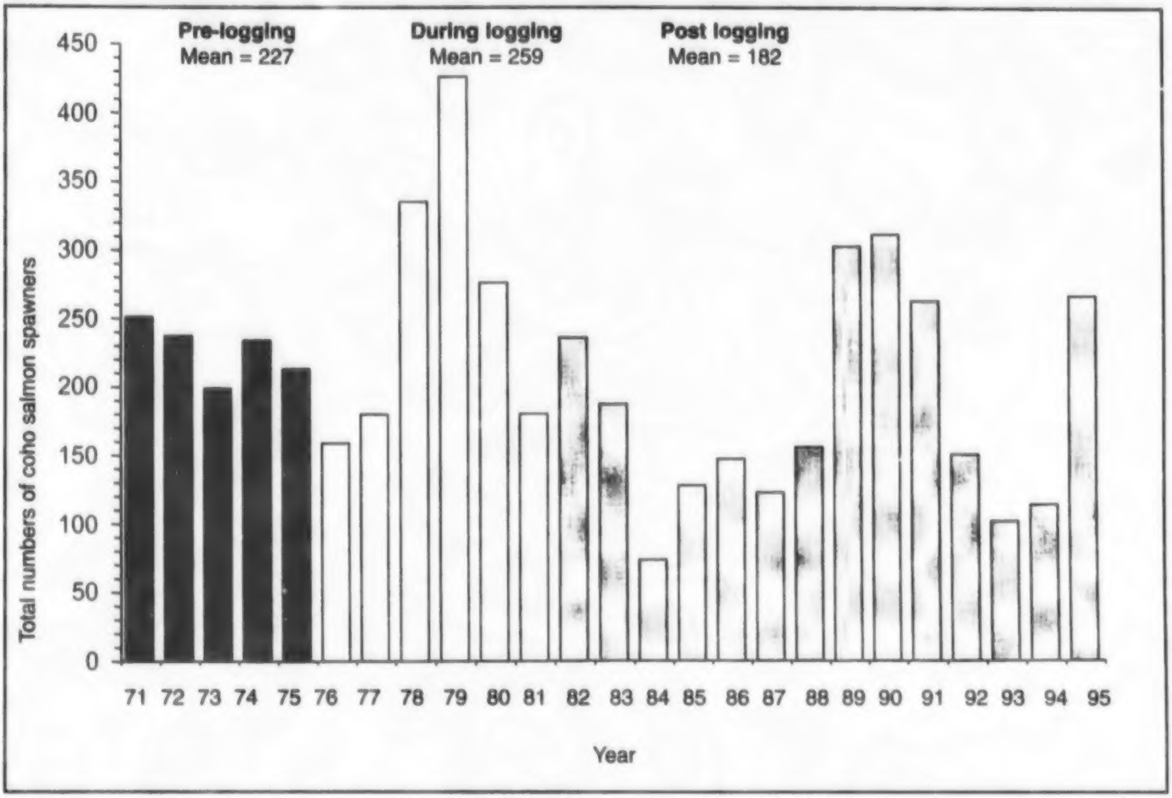

Figure 5. Annual returns of coho salmon (large adults and jacks combined) to Carnation Creek between 1971 and 1995. Mean numbers returning during the post-logging years were significantly lower than returns in the pre-logging period (Student's t, $p < 0.05$).

predator abundance (Hargreaves and Hungar 1994; Rice et al. 1995). A similar combination of conditions coincided in 1983 (Holtby and Scrivener 1989; Karinen et al. 1985) and contributed to the declines in adult coho and chum observed during the mid-1980s (Holtby and Scrivener 1989); however, the El Niño phenomenon at that time persisted for only about 1 year. In the early and mid-1990s, El Niño-like conditions prevailed for perhaps 3 years (1992–1994; Hargreaves and Hungar 1994; Rice et al. 1995). The abundances of both adult coho and chum reached historically low levels by 1994 (Figs. 2 and 3).

The cumulative effects of forest-harvesting, fishing, low ocean productivity, and elevated marine predation were associated in 1994 with the lowest recorded number of large adult coho returning to spawn in Carnation Creek. In that year, only eight fish, including just one adult female, were enumerated (Fig. 3). Carnation Creek coho thus approached year-class extinction. Low salmon returns were not unique to Carnation Creek. Similar observations were made for coho and other salmon species returning to streams throughout the west coast of Vancouver Island and elsewhere in south coastal B.C. (Hargreaves and Hungar 1994; Heizer 1991; Nelson 1993; Rice et al. 1995). Recognizing that emergency action was required for population conservation, Fisheries and Oceans Canada reduced the commercial catch of coho in the summer of 1995 (Rice et al. 1995). Reduced seasonal fishing pressure on west coast coho stocks allowed more adults to return to their spawning grounds. The overall catch of coho salmon by the fishery in south coastal B.C. under this management regime might have been lowered to roughly 50% from higher levels, which Holtby and Scrivener (1989) noted probably averaged about 67% (65–70% without historical trend).

Coho stocks appeared to respond immediately to this emergency conservation measure. Returns of large adult coho to Carnation Creek increased

dramatically from only eight in 1994 to 201 in 1995 (Fig. 3). This return exceeded the pre-logging average by nearly 22%. Coho stocks elsewhere responded similarly to reduced fishing pressure. For example, 2 300 coho were enumerated at Clemens Creek (a tributary to Henderson Lake) in the Barkley Sound region [P.J. Tschaplinski and K.D. Hyatt (Fisheries and Oceans Canada, Pacific Biological Station); unpublished project data]. Coho returns to this system for the past 10 years have generally been fewer than 200 spawners.

The strong and immediate response exhibited by Carnation Creek coho to reduced commercial fishing demonstrates the vulnerability of numerically weak salmon stocks in mixed-stock commercial fisheries. This observation also raises the question of the historical effects of fishing pressure on adult coho returns to Carnation Creek. For example, the unusually high return of 312 large adult coho in 1979 represents nearly two times the average return for the pre-logging and during-logging periods combined. One might speculate that this apparent anomaly is largely due to the majority of Carnation Creek coho escaping the fishery in the summer of 1979.

In contrast with chum salmon, coho salmon originating from a variety of sources — Carnation Creek, 281 other western Vancouver Island stocks, the Strait of Georgia, the Fraser River, and the United States — are subject to significant commercial and recreational fishing each year off the west coast of Vancouver Island (Simpson et al. 1996). The commercial troll fishery in this region is the single largest harvester of coho salmon in B.C. There is no direct measure of the number of Carnation Creek coho caught in various fisheries because smolts leaving Carnation Creek are not usually marked with coded-wire tags to study their patterns of ocean distribution and fishing mortality. The coho salmon stock from the nearby Robertson Creek hatchery (Fig. 1) is the only stock from the west coast of Vancouver Island that is tagged annually and thus provides the data required to calculate fish-harvest rates. Since 1972, fisheries have taken 54.1 to 76.7% of Robertson Creek coho without upward or downward trend (Simpson et al. 1996). The great majority (>91%) are taken off the west coast of Vancouver Island by the troll fishery, and up to about 50% are caught in or near Barkley Sound. Similar fishing rates and patterns were presumed for Carnation Creek coho salmon.

Historically, fisheries managers have used data from the Robertson Creek stock to estimate fishing and natural mortalities for west coast Vancouver Island stocks including coho salmon from Carnation Creek (Holtby and Scrivener 1989; Simpson et al. 1996). Despite the use of some untested assumptions (e.g., that hatchery and wild-stock smolts behave similarly in the ocean), fisheries scientists and managers have justified applying Robertson Creek information to estimate both fishing and natural mortalities for Carnation Creek coho because of the following observations: (1) the limited studies of coded-wire tagged fish that have been performed for the Carnation Creek stock and others have shown that catch distributions are similar among west coast Vancouver Island stocks; therefore, coho salmon from Carnation Creek and Robertson Creek should be exposed to the same fisheries for similar periods of time; (2) temporal patterns of smolt-to-adult survival are significantly correlated between Carnation Creek and Robertson Creek smolts ($r = 0.65$, $p < 0.001$) although survivals are significantly higher for Carnation Creek coho salmon (0.118 for Carnation Creek vs. 0.045 for Robertson Creek; $p < 0.001$); and, (3) temporal trends in adult escapements are generally similar between Carnation Creek coho and other west coast Vancouver Island stocks, such as those of the Stamp River system (which includes Robertson Creek) and Gold River in northwest Vancouver Island, far from Carnation Creek ($r = 0.66$, $p < 0.01$; Holtby and Scrivener 1989; Simpson et al. 1996).

Holtby and Scrivener (1989) concluded that annual fishing pressure, which was estimated to have varied between 65 and 70%, had little effect upon annual variations in adult coho salmon escapements to Carnation Creek. These investigators used a series of sequentially linked (and life-history-based) regression models to determine the relative effects of climate, forest harvesting, and fishing on the returns of adult chum and coho to Carnation Creek. The models predicted adult escapements based upon correlations between fish population responses (e.g., survival and growth) at different life stages with (a) climatic, hydrologic, and physical variables, (b) indices of freshwater habitats affected by logging, (c) realistic fishery exploitation rates varying from 0 to 0.50 for chum and 0.59 to 0.80 for coho. For their analyses, shifts in climate were determined using long-term trends in air and water temperatures from monitoring stations at Carnation Creek and other west coast Vancouver Island sites (for information prior to the start of the Carnation Creek study, and for data on ocean temperatures and sea-surface salinities; see Holtby 1988; Holtby and Scrivener 1989). Holtby (1988) partitioned the increases in stream temperatures observed after 1976 between the effects of forest-cover removal and climate change by using

multiple regression analyses. He determined that logging-associated increases in water temperatures varied from 0.7°C in December to approximately 3.3°C in August.

Simulations by Holtby and Scrivener (1989) based upon data available up to the late 1980s for Carnation Creek indicated that most of the variation in observed and predicted adult spawner returns for both chum and coho resulted from climate variations in both freshwater and marine environments (in roughly equal measure). Fishing mortality generated little change in the inter-annual patterns in adult returns associated with climate variations; however, exploitation at the highest rates resulted in 2- to 3-fold increases in interannual variation in adult numbers relative to moderate (i.e., observed) levels of fishing (Holtby and Scrivener 1989). The model predicted the collapse of salmon stocks when the effects of habitat disturbance (forest harvesting), adverse oceanic conditions, and high fishing rates coincided. This prediction appears consistent with the 1994 collapse of adult coho returns to Carnation Creek and other Vancouver Island streams.

The analyses by Holtby and Scrivener (1989) and Scrivener (1991) indicated that forest harvesting alone reduced the numbers of chum adults returning to Carnation Creek after logging by approximately 26%. However, effects upon coho were relatively minor. Less than 10% of the decline in adult coho returns by the late 1980s was predicted as a result of forest-harvest effects (Holtby and Scrivener 1989). The authors noted that their results were counter-intuitive, given that the relatively long time spent by coho in fresh water suggests that this species would be more strongly affected by forest-harvesting than chum. Conversely, chum spend more of their life cycle in marine environments and thus might be expected to be more strongly affected by marine climate shifts than by freshwater habitat changes. These results demonstrate the need for researchers and natural resource managers to be aware of biological and physical processes occurring both within watersheds and marine environments before interpreting observed patterns in salmonid production.

Other complex relationships are apparent. For example, in contrast with depressed returns of coho adults in 1994, the numbers of coho jacks returning in that year exceeded their pre-logging average (Fig. 4). The reason for this difference is unclear; however, returns of coho jacks were strongly depressed in 1993. The 1993 jacks and 1994 adults belong to the same brood year, indicating that this entire brood experienced low survival in the ocean. Simpson et al. (1996) speculated that the poor survival rates of 1993 and 1994 were due to increased predation by piscivorous marine fishes, which were abundant around Vancouver Island in both years.

Juvenile Salmonids

Chum Salmon

Forest harvesting is clearly one of several causes of the observed declines in chum salmon abundance at Carnation Creek and elsewhere along the south coast of B.C. (Holtby and Scrivener 1989; Scrivener 1991). Two-thirds of the post-logging decline of Carnation Creek chum that was attributed to forest harvesting by Holtby and Scrivener (1989) is explained by reductions in egg survival due to sedimentation of spawning and egg incubation gravels (Scrivener 1991). Observed egg-to-fry survival for chum has declined by about one-half from a mean of 20.3% in pre-logging years to 10.9% after logging (Hartman and Scrivener 1990).

Most chum at Carnation Creek spawn in the lowermost portion of the system located downstream of the main fish weir. Between 68 and >99% of all chum spawn in this area, all of which is under tidal influence, and most of which is regularly inundated with saline water (Tschaplinski 1982b; 1988). Most of the remaining spawners usually migrate only short distances (e.g., 100 m) upstream of the weir. Frozen-core gravel samples have shown that all of this area used by chum has been subjected to increases in fine sediment deposition after forest harvesting (Hartman and Scrivener 1990; Scrivener 1988a, 1988b, 1988c, 1991; Scrivener and Brownlee 1982, 1989).

Much of the sediment added to the stream came from eroding banks in areas upstream where both careful and intensive streamside forest-harvest treatments were applied (Hartman et al. 1987; Hartman and Scrivener 1990). During-logging and after-logging (1978–1985) bank erosion accelerated in these clearcut areas in association with increased frequencies and magnitudes of seasonal freshets (Hartman et al. 1987; Scrivener 1988b, 1988c; Scrivener and Brownlee 1989). Freshets transported the eroded materials downstream into the leave-strip treatment including the sites used by chum salmon (Hartman and Scrivener 1990; Scrivener 1988b). Analyses of frozen-core gravel samples showed that most of the material that reached those spawning sites and accumulated in the streambed consisted of sands, which increased at depths where chum salmon eggs would occur (in the middle and deep layers of the cores representing streambed depths of 12 to 35 cm; Scrivener

Introduction

Despite the extensiveness of forestry in Finland, the number of studies conducted on the effects of forestry on fish and fisheries in running waters are limited. Some studies have been made on the effects of forest drainage on ascending behavior of salmonid fish, on smolt production, and on the filling of spawning grounds with sand (e.g., Viitala and Hyvärinen 1986; Kännö 1981). A few studies about the effects of forestry on fish and fisheries were done in Scandinavia as well (e.g., Bergquist et al. 1984; Simonsson 1987). Most studies on this subject have been done in North America. However, North American results do not necessarily apply to Finnish circumstances because climate, soil, geomorphology, tree species, harvesting practices, characteristics of watercourses, and fish stocks are different.

In this study, we examined the effects of some forestry activities on brown trout (*Salmo trutta L.*) densities in a basin where the proportion of forests and bogs in the total catchment area was high. The study was done in the River Isojoki basin, western Finland, which has a remarkable value both for fishery and for the conservation of the biodiversity of the brown trout stocks. The questions in the study were: 1) Could trout density of the brooks be explained by environmental factors? 2) What might these factors be? and 3) Can these factors be considered as forestry factors or as factors that were altered by forestry?

Study Area

The River Isojoki is located in the western Finland (62°N, 22°E), and it flows to the Gulf of Bothnia in the northern part of the Baltic Sea. The catchment area of the river is 1117 km^2. About one third of this area was included in this study. The main stream of the river is about 75 km long, and it has three main tributaries. One of the tributaries and its catchment (River Kärjenjoki area) was excluded, because it is known that no natural trout stocks exist there. A typical characteristic of the River Isojoki system is a large number of brooks (over 50), compared with other Finnish river systems of the same size.

The main land use of the study area (forests and bogs) was forestry, at about 97%. The share of agricultural use was about 2%. Besides logging and harvesting, forestry in the study area includes forest improvement. The most pronounced improvement method during the last three to four decades has been forest drainage. Drainage is used to enhance tree growth by improving moisture conditions because the inclination gradients are low. In the study area a typical gradient is 3–4 m/1 km. The mean abundance of ditches was about 10 km/km^2. Dredging of the brooks has been a drainage method used in connection with ditch excavating. Dredging involves excavating the brooks to be deeper, wider, or straighter to carry water from the ditches more effectively. Dredgings were carried out during the 1960s and 1970s.

Material and Methods

The brooks of the River Isojoki were surveyed by electrofishing to estimate (Bohlin et al. 1989) brown trout density, the dependent variable in the analyses. The sample site locations were selected on the basis of maximum relative silvicultural and forestry land use of the upper catchment, and according to the existence of potential brown trout habitats, i.e., rapids. The number of sample sites in one brook varied from one to five. The minimum distance between sites were about 500 m. In the analysis there were 58 sampled sites representing 41 different brooks. Young-of-the-year parr were excluded from the analysis, because of low or sporadic catchability. Brown trout was, in practice, the only fish species obtained in the catches.

The sampled area, mean depth, mean surface velocity, conductivity, and pH value of the water were measured. The distribution of bottom material [sand, gravel, stones (<2 cm, 2–10 cm, 10–30 cm, >30 cm), and clay], distribution of surface velocity (<0.2 m/s, 0.2–0.4 m/s, 0.4–0.7 m/s, >0.7 m/s), coverage of bottom vegetation, and shading of trees in the site were estimated visually. Also, the abundance of bottom pools (minimum depth 30 cm, minimum diameter 50 cm), undercut banks (minimum depth 20 cm, minimum length 100 cm), and stream cascades (minimum fall 15 cm) were estimated visually. The sample sites were classified as either dredged or natural. There were in some cases both dredged and natural sample sites in the same brook.

A cartographic survey was made to measure the area of upper catchment for each sample site, the land use or land type (e.g., forest, bog, and agriculture) and total length of ditches in the catchment.

A multivariate regression model was used to analyze the variation in brown trout densities. The independent variables were first nominated on the basis of correlation analyses. After that, the potential independent variables were selected for the final regression models. To linearize the variables and to normalize the distributions, the variables were $\arcsin\sqrt{(x)}$ or $\log(x)$-transformed. The normality,

independency, equality of variances, and autocorrelation of variables or residuals were controlled. Statistical analyses were calculated with SYSTAT (1992) programs.

Results

In a regression model for all fishing sites (Table 1), the brown trout density was determined with four significant variables ($p \leq 0.05$). The variables were the abundance of pools, abundance of undercut banks, pH value, and relative total length of ditches in the upper catchment. The last one was the only negatively regressioned variable. These four variables determined 42% of the variation in brown trout densities.

Trout density in natural sites (mean 11 trout/100 m^2) was clearly higher than in the dredged sites (4.7 trout/100 m^2). The log-transformed means differed significantly ($|t| = 2.166$, DF = 74, $p = 0.03$). Therefore, sample site results from natural and dredged sites were also analyzed separately.

Table 1. Regression model of brown trout density [log(x+1)] of all sample sites in streams. N = 58, F = 9.795, $p<0.01$, $R^2 = 0.42$, D = 1.946 = Durbin–Watson-value for independency of residuals, r = 0.001 = autocorrelation.

Independent	Coefficient	Standard error	Standard coefficient	T	P
Constant	−1.371	1.076	0.000	−1.274	0.21
Pools	1.022	0.261	0.414	3.919	<0.01
pH value	0.444	0.157	0.308	2.834	0.01
Undercut banks	0.594	0.254	0.258	2.343	0.02
Upper ditches	−0.494	0.250	-0.217	−1.979	0.05

Table 2. Regression model of brown trout density [log(x+1)] from sample sites in natural streams. N = 19, F = 22.47, $p<0.001$, $R^2 = 0.90$, D = 2.111 = Durbin–Watson-value for independency of residuals, r = −0.144 = autocorrelation.

Independent	Coefficient	Standard error	Standard coefficient	T	P
Constant	2.081	1.590	0.000	1.309	0.21
Pools	1.064	0.262	0.452	4.056	<0.01
Proportion of bog	−3.372	0.907	−0.337	−3.716	<0.01
Shading	−2.151	0.759	−0.307	−2.834	0.01
Mean depth	0.613	0.247	0.230	2.485	0.03
pH value	0.289	0.127	0.212	2.265	0.04

Table 3. Regression model of brown trout density [log(x+1)] from sample sites in dredged streams. N = 40, F = 4.732, p = 0.01, $R^2 = 0.28$, D = 1.694 = Durbin–Watson-value for independency of residuals, r = 0.129 = autocorrelation.

Independent	Coefficient	Standard error	Standard coefficient	T	P
Constant	−2.169	1.424	0.000	−1.522	0.14
Undercut banks	0.770	0.294	0.380	2.623	0.01
pH value	0.422	0.218	0.278	1.933	0.06
Pools	0.534	0.333	0.229	1.605	0.12

smolts were enumerated at the main weir in spring (Fig. 6). No statistically significant changes in the numbers of cutthroat trout smolts produced from Carnation Creek are apparent after logging. The pre-logging mean of 42 smolts is nearly identical to the post-logging mean of 37 (Fig. 6; Student's t, $p > 0.05$). The trend to higher average numbers in the during-logging period is not significant due to high variability among years ($p > 0.05$).

The abundance of steelhead trout smolts has declined on average at Carnation Creek from 246 before logging to only 85 between 1982 and 1995 (Fig. 6). Trout smolt numbers have thus fallen to <35% of their pre-logging levels; however, variability among years has been so high that these trends are not statistically significant (Student's t, $p > 0.05$). Sharp reductions in some years are nevertheless apparent after 1978 when freshet-associated changes in the stream channel and consequent loss of rearing habitats were first observed in clearcut areas of the stream (Hartman et al. 1987; Hartman and Scrivener 1990).

Steelhead trout may be more susceptible to main-channel habitat loss than either juvenile coho or cutthroat trout, especially in winter when freshets are common. Rainbow in Carnation Creek are restricted to main-channel habitats in contrast with coho and cutthroat trout, which also occupy tributaries, especially in winter (Brown 1987; Tschaplinski and Hartman 1983). During winter, many juvenile coho and cutthroat (at least 20% of all age 0 fish) seek shelter from scouring freshets by inhabiting off-channel sites including tributaries (Brown 1987; Hartman et al. 1996; Tschaplinski and Hartman 1983). On the other hand, rainbow seek shelter in main-channel pools and undercut banks associated with logs and tree roots (Bustard and Narver 1975).

In some years after logging (e.g., 1984), low abundance of steelhead trout smolts in spring occurred after winters with frequent severe freshets (Fig. 6). With the loss of main-channel shelter habitats in clearcut sections of Carnation Creek (Hartman et al. 1987; Hartman and Scrivener 1990), the salmonid mortality associated with freshets was likely more pronounced in post-logging years. By comparison, winters without strong freshets were sometimes associated with high numbers of rainbow smolts in the following spring, even after logging. For example, the abundance of rainbow trout smolts in the spring of 1993 is the third highest on record (Fig. 6) and occurred after a relatively mild winter without strong freshets (unpublished project data). Despite these observations, patterns for rainbow trout are not consistent among years, and are obscured by generally low abundance of this species in Carnation Creek. The effects of forest harvesting treatments on freshwater rearing habitats and fish populations are more readily apparent for the relatively abundant coho salmon.

Coho Salmon

The decline in the numbers of adult coho salmon returning to Carnation Creek after forest harvesting might be partly explained if the capacity of the stream to support populations of juvenile coho has decreased. Consistent with this notion, significant post-logging declines in the abundance of juvenile coho rearing in the stream during summer have been observed (Fig. 7; Student's t, $p < 0.05$). Before forest harvesting, the freshwater habitats in Carnation Creek supported 11 944 ± 2 117 coho juveniles in late summer (Fig. 7; late September to early October, fry and yearlings combined). Between 1976 and 1981, when most of the valley bottom of the main basin was harvested, late-summer coho populations increased to 13 656 ± 5 661, but this increase was not statistically significant (Fig. 7; Student's t, $p > 0.05$). However, the total numbers of coho fry and yearlings rearing in Carnation Creek since 1982 have fallen by approximately one-half to 6 127 ± 1 684 (Fig. 7; Student's t, $p < 0.05$).

This post-logging reduction in juvenile coho abundance is also partly due to marine survival variations that have reduced the numbers of adults returning to spawn (Hartman and Scrivener 1990; Holtby and Scrivener 1989; Scrivener and Andersen 1984). The decline in the numbers of adult females (approximately 34%) has resulted in fewer eggs deposited into the streambed, and consequently, fewer fry have inhabited the system during summer since 1982. If there were no significant post-logging reductions in the amount of suitable rearing habitat, reduced numbers of fry inhabiting the stream after logging would reflect a system supporting fry at numbers well below capacity due to insufficient spawners. However, several observations indicate that the post-logging reduction in the abundance of juvenile coho in Carnation Creek during summer is also the result of logging-caused reductions in egg-to-fry survival and the quantity and quality of summer rearing habitats (Hartman and Scrivener 1990; Hartman et al. 1996).

After forest harvesting, egg-to-fry survival declined by about one-half in Carnation Creek, from 28.8 to 15.6% (Hartman and Scrivener 1990). This trend was similar to that shown by chum salmon,

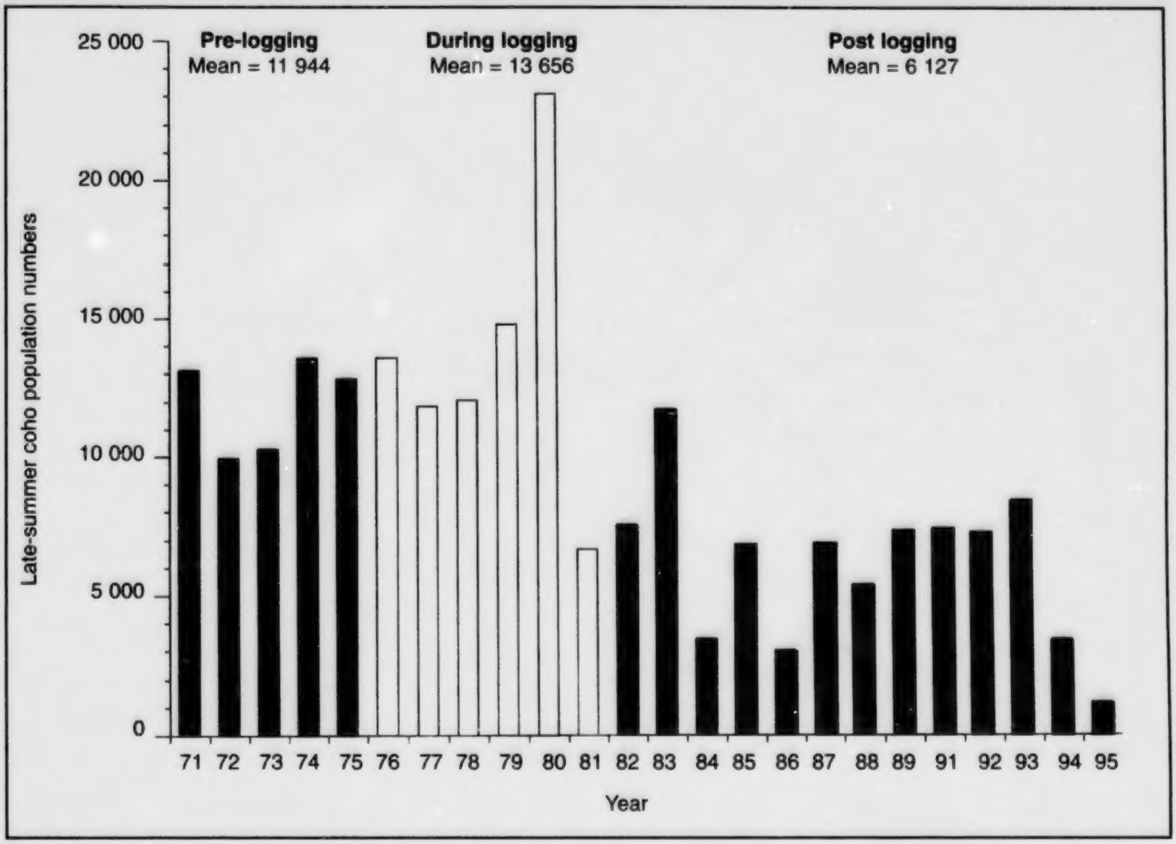

Figure 7. Abundance of juvenile coho salmon rearing in Carnation Creek in late summer (September–October). These populations consisting mainly of fry plus some yearlings have declined by about one-half between pre-logging and post-logging periods (Student' s t, $p < 0.05$). No data are available for 1990.

which was also in part associated with increased percentages of sand and pea-sized gravel (by 5.7 and 4.6%, respectively) in the streambed in the leave-strip area downstream of the clearcut portions of the creek ($r = 0.81$, $p < 0.001$; Scrivener and Brownlee 1989). However, reductions in coho egg-to-fry survival were primarily associated with increased rates of stream-bank and channel erosion after logging in both the intensively harvested and carefully harvested clearcut treatments (Toews and Moore 1982; Scrivener and Brownlee 1989). The proximal cause of increased mortality in coho embryos was increased streambed scour and deposition during freshets after logging (Holtby and Scrivener 1989).

Reduced fry abundance at Carnation Creek after logging is thus the combined result of these processes in fresh water plus reduced marine survival that resulted in fewer spawners returning to the creek. Additionally, the capacity of the stream to support those fry that survived to emerge from the streambed in spring also declined after logging. Most of this decline can be attributed to stream morphology changes that have occurred both in the intensively harvested and carefully harvested clearcut areas. Prior to 1982, surveys of fish abundance showed that Carnation Creek was able to support as many as 23 095 juvenile coho at the end of summer (Fig. 7; including 20 953 fry). This peak in juvenile abundance observed in 1980 exceeded mean population sizes in the pre-logging and during-logging periods by >1.9- and 1.7-fold, respectively, and was produced from the 25-year peak spawner return in the autumn of 1979 (312 large adults including 176 females). The greatly elevated number of juveniles rearing in Carnation Creek in 1980 suggests that habitat quantity in the stream did not limit the summer abundance of juveniles in most years prior to the post-logging

The Upper Little Smoky: Integrating Timber Harvesting with Fisheries and Recreation Management - A Case Study

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Abstract

The upper Little Smoky River has long been considered a popular sport fishing and wilderness recreation area. Alberta Newsprint Company (ANC) is proposing an integrated approach to allow timber harvesting to occur without compromising the integrity of the fisheries or the capacity of the area to provide an outdoor recreation experience. The paper presented would provide an opportunity to review ANC's plan and will hopefully generate thought and discussion on the potential for integrating industrial activity, recreation use, and fishereis habitat. Alberta Newsprint Company's intention is the integrate stakeholder concerns into the plan as much as possible with the plan having a firm basis in scientific principles. It is expected that the plan will include a variety of management scenarios ranging from special logging techniques to special access management considerations. The plan will also include an assessment program to determine if these special practices are having the affect that was intended.

Introduction

As part of a commitment to sustainable forest management, Alberta Newsprint Company (ANC) is developing a management strategy for the Little Smoky River area. The Little Smoky's various resource values include timber, caribou, wildland recreation, sport fisheries, and oil and gas reserves. Many recent changes have led to a need for a comprehensive planning approach for key areas within Alberta. These changes include increased resource consumption, more pressure on the land base for non-consumptive type uses, increased desire by the public to be involved in land use decisions, and increased awareness of the needs of certain fish and wildlife species. Provincial government reorganization has also created changes in how land use decisions are made and by whom. The approach used by ANC is to integrate these diverse resource needs in order to maximize all stakeholders' benefits.

The Little Smoky area falls within ANC's Forest Management Agreement (FMA) area. The FMA is 381 000 ha in size and supports a net annual allowable cut (AAC) of 570 193 m^3 comprising 66.4% of the coniferous AAC and 100% of the deciduous. The balance of the coniferous fiber rights are held by three other forest products companies, namely Millar Western, West Fraser, and Mostowich Lumber.

Branton, G. 1998. The upper Little Smoky: integrating timber harvesting with fisheries and recreation management—a case study. Pages 73–76 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

2 213 ± 424 smolts were enumerated annually at the main fish weir. Early in the during-logging period (1978), these numbers had nearly doubled to 4 246 compared with the pre-logging mean (Fig. 8). Annual smolt abundance increased nearly 1.7-fold to 3 688 ± 813 in the 6 years during logging. Numbers have remained high to the present. A 25-year peak migration of 5 253 coho salmon smolts occurred in 1992, exceeding the pre-logging mean by nearly 2.4-fold. Mean annual smolt abundance in the post-logging period has been 3 441 ± 495 (Fig. 8). Therefore, in the post-logging period, Carnation Creek fry, at roughly one-half their pre-logging abundance, have produced 55% more smolts. Smolt biomass has also increased after logging. The average weight of both age-1 and age-2 smolts has increased since 1977; for example, the mean weight of age-1 smolts increased by 1.5 g or one-third of their mean weight prior to logging (Holtby 1988). The reason for these phenomena is that coho fry rearing in the stream in late summer are surviving the winter at higher rates after logging due to the temperature-related effects of forest harvesting (and climatic warming) on fry emergence timing and seasonal growth (Hartman et al. 1990; Holtby 1988; Scrivener 1988c).

The multiple regression analyses that Holtby (1988) used to determine the effects of increasing water temperature on chum salmon, and partition the effects due to climate change and forest harvesting, were also applied to coho salmon. He demonstrated that the same logging-associated increases in winter stream temperatures that allowed chum salmon eggs to develop more rapidly, and chum fry to emerge earlier in spring, have had similar effects upon coho. During and after logging, coho fry were emerging from the streambed up to 6 weeks earlier in spring (Holtby 1988). Earlier emergence provided a period of summer growth for these fish that was as much as 6 weeks longer than in pre-logging years. Additionally, lower numbers of fry rearing in Carnation Creek after logging resulted in increased growth rates due to density-dependent reductions in competition for food (Holtby 1988; Scrivener and Andersen 1984). Consequently, after logging, coho fry grew an average of 11 mm longer by the end of their first summer compared with growth in pre-logging years (Hartman and Scrivener 1990). [For the same reasons, trout fry increased in mean length by 18 mm after logging in Carnation Creek (Hartman and Scrivener 1990).] This larger size was positively associated with improved overwinter survival in the stream after logging (ANOVA, $r = 0.91$, $p < 0.001$; Holtby 1988). Larger coho were apparently better able to survive winter conditions that include frequent scouring freshets (Brown and McMahon 1988; Tschaplinski and Hartman 1983).

Increased seasonal growth resulting primarily from post-logging increases in water temperatures has also radically changed the age structure of coho smolt populations (Fig. 9). Prior to forest harvesting, nearly one-half of all coho rearing in Carnation Creek required 2 years of growth before they were large enough to transform into smolts and migrate seaward (Fig. 9; 1971–1975). During and after logging, increased growth resulted in age-2 fish becoming relatively rare (Fig. 9).

Increases in stream temperatures were observed almost immediately after clearcut harvesting removed significant portions of the streamside forest canopy (Hartman and Scrivener 1990; Holtby 1988). Coinciding with this rapid temperature shift, the numbers of coho fry able to achieve smolt size after rearing for only 1 year in the stream increased dramatically in 1976, the first year of forest harvesting (Fig. 9). The proportion of age-1 smolts increased significantly from 55.3 ± 13.8% in the pre-logging period to 76.7 ± 11.0 in the during-logging period (Student's t, $p < 0.05$). This proportion increased further to 91.4 ± 3.2% in the post-logging period (1982–1985; Student's t, $p < 0.05$). Therefore, the temperature-related effects of forest harvesting upon juvenile coho growth and age structure that were established soon after streamside harvesting began, persist in the watershed 20 years later. These effects will likely continue for several years until a new riparian forest canopy is established at Carnation Creek, and water temperatures and fish growth both decline toward pre-logging levels.

Although smolt numbers have increased after logging, reductions in their marine survival are implied from the declining numbers of adults returning to Carnation Creek since 1982 (Fig. 3). The marine survival of coho smolts from Carnation Creek has decreased steadily since the 1970s, and the lowest survivals have occurred in the most recent years for which these data are available (Fig. 10). Although larger size has allowed coho fry to survive the winter better in fresh water after logging, similar relationships have not been observed for coho smolts in the ocean (Hartman and Scrivener 1990; Holtby et al. 1990). The marine survival of coho smolts appears unrelated to (1) the size of either age-1 or age-2 smolts, or (2) smolt age (Fig. 11; paired t-tests, all $p > 0.05$; Holtby et al. 1990). After logging, the mean size of coho smolts migrating seaward has actually declined because most smolts are now age-1

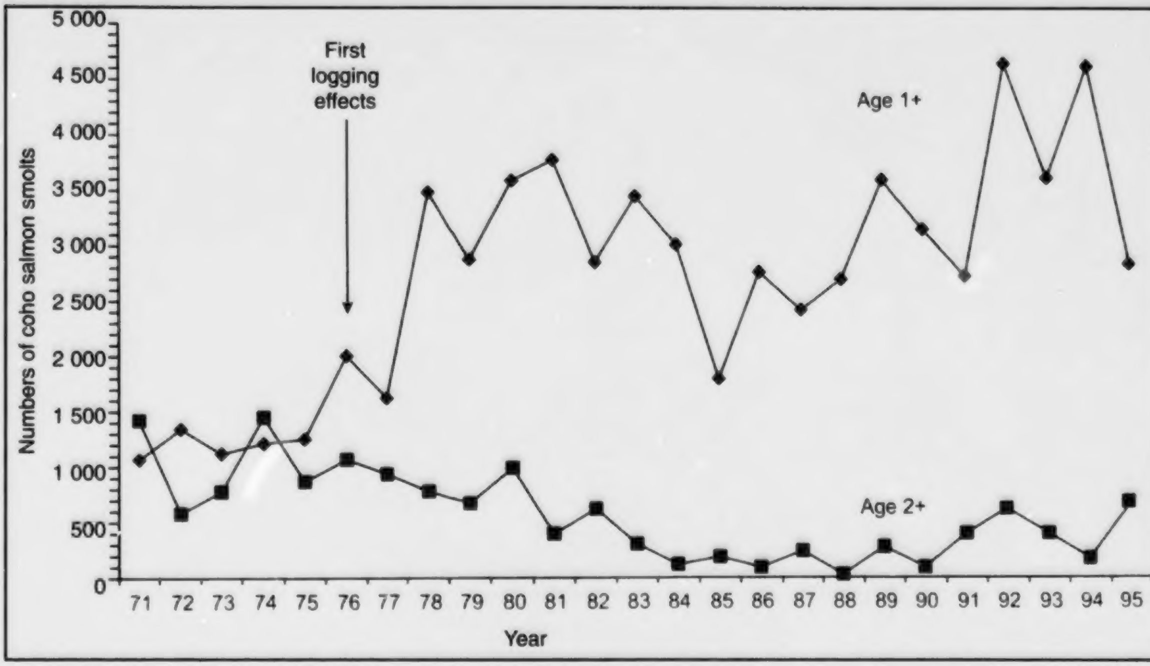

Figure 9. Age composition of coho salmon smolts migrating from Carnation Creek each spring between 1971 and 1995. Since 1982, >91% of coho smolts are age-1 fish.

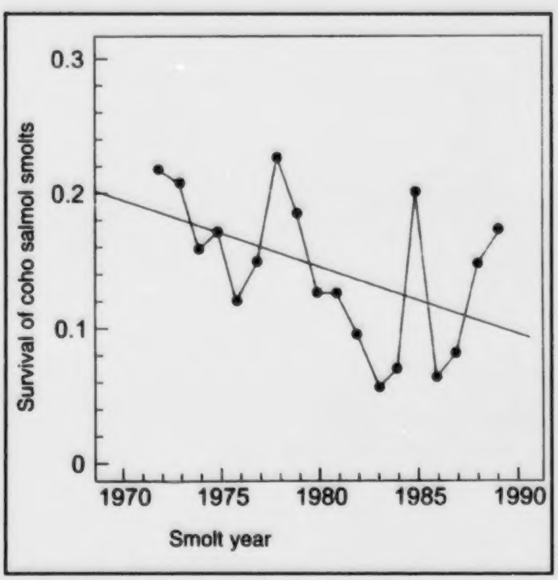

Figure 10. Marine survival of coho salmon smolts from the early 1970s to the late 1980s. Data on adult returns suggest that this significant ($p < 0.05$) long-term decline in survival continues in recent years. (Figure provided by L.B. Holtby, Fisheries and Oceans Canada, Pacific Biological Station, Nanaimo, B.C.)

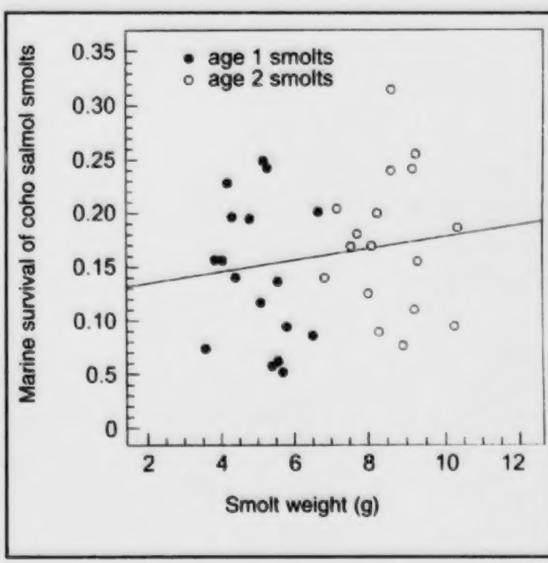

Figure 11. Marine survival versus size and freshwater age of coho salmon smolts from Carnation Creek. Survival was not significantly correlated with smolt age or weight (i.e., body size) within or between the two age groups (paired t-tests, all $p > 0.05$; from Holtby et al. 1990).

Backcountry Recreation

Fundamental to the development of this strategy will be an assessment of backcountry wildland recreation opportunities. These will include: view scape assessments, unique geologic and historical features assessments, potential campsite locations, and human travel corridors, including roads, trails, and waterways. The public involvement component of the strategy development will be critical in capturing the wants and desires of those wishing to utilize the recreational opportunities available in the area.

Sustainable Wood Supply

Timber-harvesting commitments have been made in the Little Smoky River area by the Alberta government. These commitments will be met by using a variety of innovative methods. The fundamental premise in the timber-harvest planning and approaches used will be to follow natural disturbance regimes as closely as possible. Wildfire has historically been the main influencing natural disturbance. Historically, wildfires vary in size, location, intensity, and reoccurrence cycles. Attempts will be made to understand these variations and to harvest the timber in a way that comes as close as possible to replicating fires across the landscape. To date, a study has been established that looks at how lichen regeneration was affected in a large harvested area and the effect on caribou–predator relationships of a large cut over. A second study is under development that will look at selective harvesting and commercial thinning in lodgepole pine in an attempt to minimize visual disturbance and replicate a less intense wildfire.

Due to issues raised by local fishers and outdoor recreationists, efforts will be made to minimize the visual impact of timber harvesting. Such things as protection of roadside vegetation, small blocks, and non-clear-up systems will be considered in areas where there is potential for high recreation use.

Access

Development of a road network is critical for resource extraction, forest renewal, forest protection, and some recreation uses. The amount and type of roads developed, however, will have a marked effect on the inherent wilderness characteristics of this area. How roads are planned and constructed will be a very important aspect of the strategy. The first step proposed is identification of road corridors. These corridors will be identified on maps as locations for road construction when and if they are ever needed. Decision criteria for where roads should go will be based on all resource values. By discussing, preplanning, and integrating all concerns well in advance of actual road construction, the potential for conflict will be much reduced. The success of this approach will hinge on a commitment from all parties to develop the road plan together and then to follow it once it is complete.

Road-use controls will also be considered. Timing restrictions, for instance, to minimize sensory disturbance to wildlife during critical periods will be implemented where appropriate. Road closures will also occur both on a temporary, seasonal basis and once a specific road is no longer needed. Roads will be built to a minimum standard in terms of width, in the interest of minimizing disturbance while ensuring environmental integrity and user safety.

Fishery Protection

The Little Smoky River and its tributaries within the study area will be protected by a variety of means. In terms of harvesting activities, ANC Timber Ltd.'s operating ground rules define specific operating restrictions to ensure protection of watercourses.

In addition to ground rule controls, ANC is participating in research work to improve the level of understanding of the fisheries within the Little Smoky drainage. Studies are currently underway to determine the timing and extent of spawning migration of Arctic grayling. The results of this work will be used to direct activities within the Little Smoky area in a way that will not negatively affect grayling.

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Hydrogeology of brook trout (*Salvelinus fontinalis*) spawning and incubation habitats: implications for forestry and land use development

R. Allen Curry and Kevin J. Devito

Abstract: We demonstrated that nearshore spawning and incubation habitats of brook trout (*Salvelinus fontinalis*) are manifestations of lenses of coarse overburden materials underlying the nearshore zone. Lenses directed and accelerated groundwater flow into the habitats. They were <17 m wide, >1 m thick, and could be restricted to the nearshore zone or extend at least 20 m into the terrestrial catchment. Recharge areas necessary to sustain discharge in the habitats were estimated to encompass 3–10 ha, or 1–97%, of the associated terrestrial catchment. A 90-m buffer zone adjacent to the shoreline protected only 9–55% of the required recharge area. A hydrological approach to defining habitat protection measures is suggested.

Résumé : Nous avons démontré que les habitats de frai et d'incubation de l'omble de fontaine (*Salvelinus fontinalis*) sont des fonds où se retrouvent des accumulations lenticulaires de matériaux bruts, près des rives. Les lentilles de matériaux bruts dirigaient et accéléraient le débit de l'eau dans les habitats. Elles avaient une largeur de <17 m, une épaisseur de >1 m et se retrouvaient soit près des rives, ou jusqu'à 20 m à l'intérieur de la zone terrestre du bassin hydrographique. Nous avons estimé les aires de recharge nécessaires au maintien de la décharge comme étant de 3 à 10 ha, soit 1 à 97% du bassin hydrographique terrestre associé. Une zone tampon de 90 m adjacente aux rives protégeait seulement 9 à 55% de l'aire de recharge requise. Nous suggérons une approche hydrologique pour définir les mesures de protection de l'habitat.

[Traduit par la Rédaction]

Introduction

The spawning and incubation of brook trout (*Salvelinus fontinalis*) in Canadian Shield water requires areas of distinct and constant groundwater discharge located in the nearshore zone (Curry and Noakes 1995; Curry et al. 1995). Known sites for reproduction within a water body are rare (<3), small in area (<10 m^2), and composed of unconsolidated, cobble–gravel–sand complexes (Snucins et al. 1992) created by glaciofluvial activities 10 000 – 15 000 years ago in this region (Price 1973).

The presence of shallow, unconsolidated, permeable materials overlying metamorphic bedrock suggests that groundwater is derived from local sources (Toth 1963). Local groundwater generated in the terrestrial catchment flows towards the lower potentials in the nearshore zone at a rate controlled by the hydraulic gradient and material permeability (Freeze and Cherry 1979). The greatest rate of discharge to surface water bodies occurs in the nearshore zone (Winter 1974). Given a uniform composition of overburden in the watershed, the rate of groundwater discharge would be similarly uniform across all nearshore areas. However, rates are extremely variable (e.g., Shaw and Prepas 1989) and they are significantly greater in brook trout reproductive habitats than in nearshore areas not selected by trout (Curry and Noakes 1995).

Despite the importance of groundwater discharge for successful brook trout reproduction, there is no information on the hydrological linkage between the required nearshore habitats and catchment where groundwater originates. The objective of this study was to increase our understanding of the interaction between catchment hydrogeology and brook trout spawning and incubation habitats. We hypothesized that lenses of coarse overburden materials accelerated the groundwater flow observed in the nearshore habitats. In addition, discharge rates were used to estimate the area of recharge necessary to sustain groundwater flow to the nearshore habitats and provide an evaluation of buffer zones as protection for brook trout spawning and incubation habitats.

Methods

Study sites

Single brook trout spawning and incubation sites in two lakes and one stream on the Precambrian Shield in central Ontario were examined from September to May 1990–1992. The

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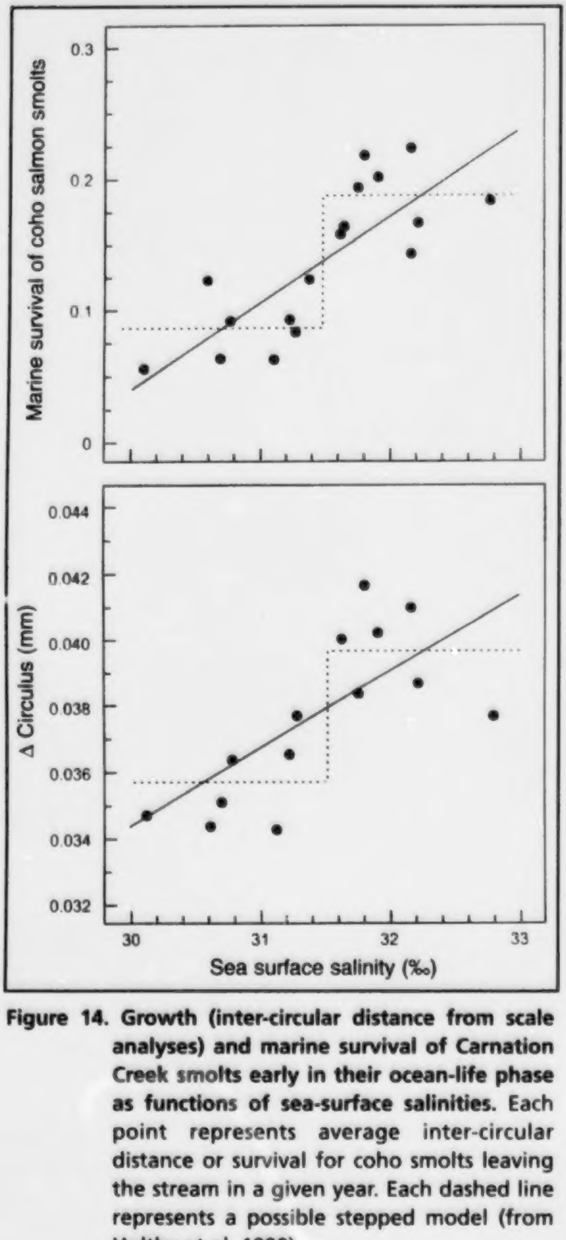


Figure 14. Growth (inter-circular distance from scale analyses) and marine survival of Carnation Creek smolts early in their ocean-life phase as functions of sea-surface salinities. Each point represents average inter-circular distance or survival for coho smolts leaving the stream in a given year. Each dashed line represents a possible stepped model (from Holtby et al. 1990).

during winter-like conditions less favorable for growth and survival.

Notwithstanding the effects of forest harvesting, coho smolt survival in the past several years has clearly been further depressed by physical-regime shifts in the ocean. In years when ocean surface temperatures are high, and salinities are low (e.g., El Niño years), the interface between the Alaskan and Californian current systems (the subarctic boundary) moves northward and away from the west coast of Vancouver Island. This zone is displaced because of a northward shift of warm, nutrient-poor water that suppresses seasonal upwelling, depresses ocean productivity, and is associated with low rates of survival for juvenile salmonids and other species such as herring (*Clupea harrengus pallasi*; Holtby et al. 1990). At the same time, large numbers of predators, mainly chub mackerel (*Scomber japonicus*) and Pacific hake (*Merluccius productus*), move northward into the coastal waters of B.C., and are believed to consume large numbers of juvenile salmonids including coho (Fulton and LeBrasseur 1985; Holtby et al. 1990).

Ware and McFarlane (1988) concluded that changes in the abundance of herring off Barkley Sound are primarily due to changes in the intensity of predation. They have noted that the biomass of piscivorous predators, such as Pacific hake, have been sufficiently high to account for all of the annual mortality within herring stocks in some years. Holtby et al. (1990) have shown that the survival of coho smolts and 1-year-old and 2-year-old herring are co-variant (Fig. 16; ANOVA, $r = 0.6$, $p < 0.01$). They noted that coho smolts and herring between 1 and 2 years old are similar in size, have overlapping diets, and occur together in both Barkley Sound and other rearing areas in the coastal waters of western Vancouver Island. Although Pacific hake and chub mackerel prey primarily upon herring, which usually greatly outnumber coho, even incidental predation upon coho can cause substantial mortality in coho population, including coho from Carnation Creek. Predation probably increases in years when ocean conditions, caused by El Niño or something similar, slow salmon growth and keep juvenile coho small and susceptible to predators (Holtby et al. 1990). Therefore, Holtby et al. (1990) suggested that most of the reduction in the ocean survival of coho smolts in recent years was the result of predation as the ultimate consequence of low ocean productivity. Holtby and Scrivener (1989) concluded that this ocean process was the largest determinant of declines in survival of coho (and chum) salmon from Carnation Creek and elsewhere from the west coast of Vancouver Island (Fig. 12).

The effects of forest harvesting upon Carnation Creek coho and chum salmon have been relatively small (for each species, explaining respectively about 26% and <10% of the post-logging variability in the abundance of adult returns) when compared to the effects of climate change in both fresh water and

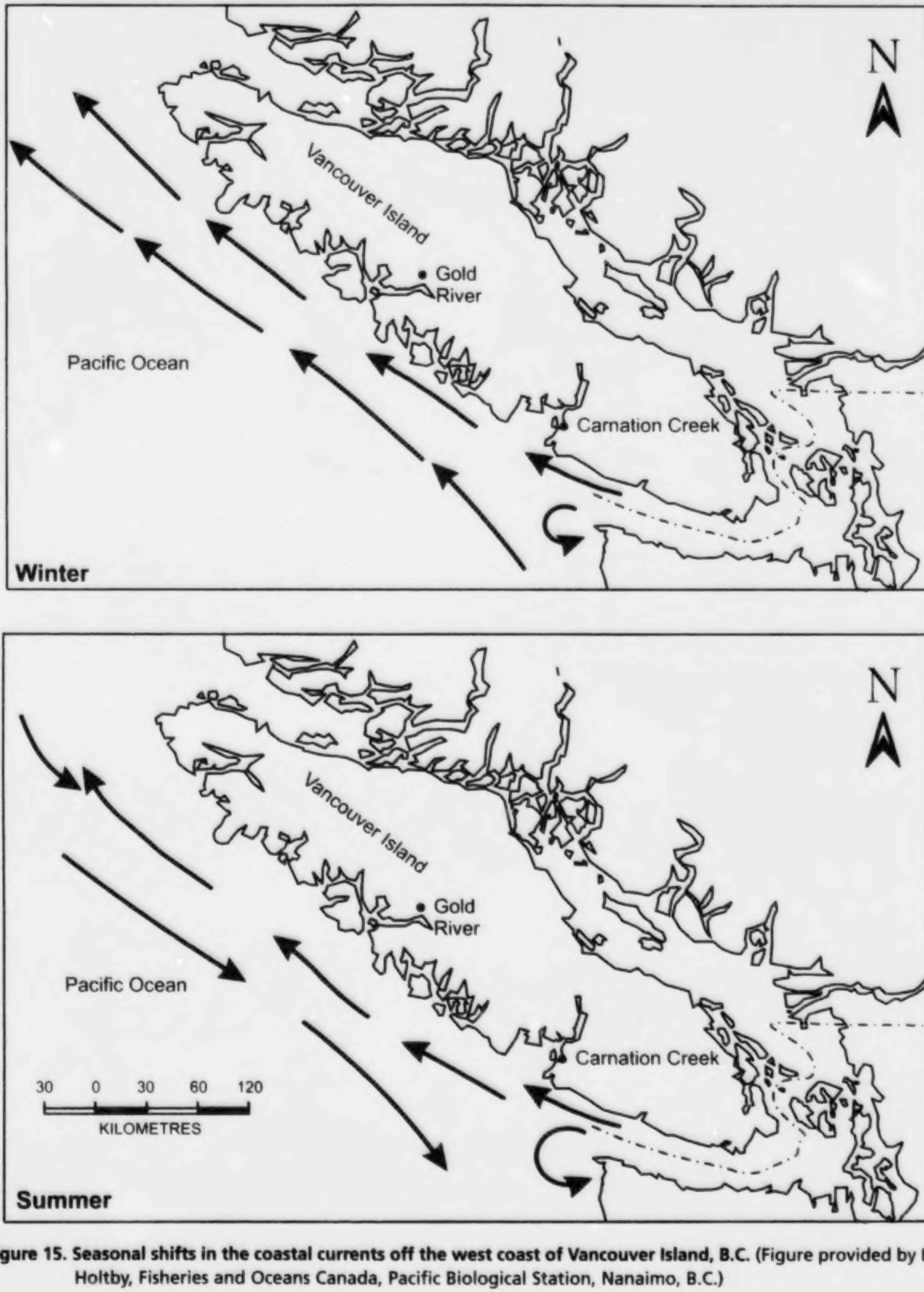


Figure 15. Seasonal shifts in the coastal currents off the west coast of Vancouver Island, B.C. (Figure provided by L.B. Holtby, Fisheries and Oceans Canada, Pacific Biological Station, Nanaimo, B.C.)

Fig. 5. Hydraulic head above the lake surface (*a*) and hydraulic conductivities (*b*) along the shoreline and through the reproductive habitat at Dickson Lake.

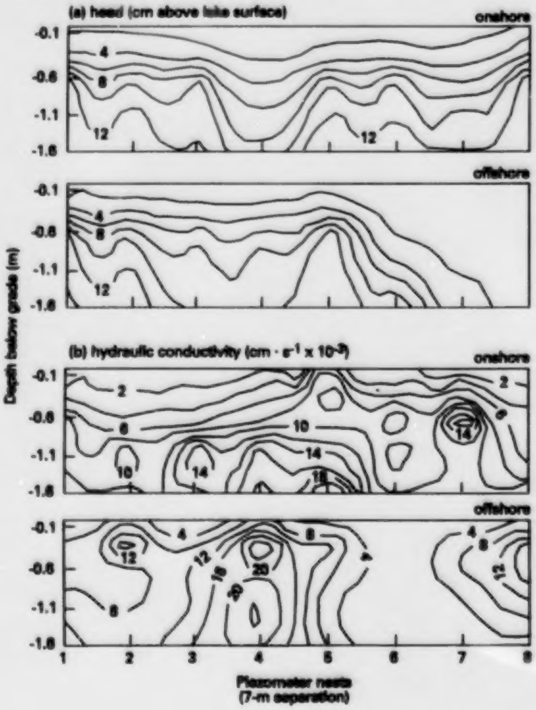
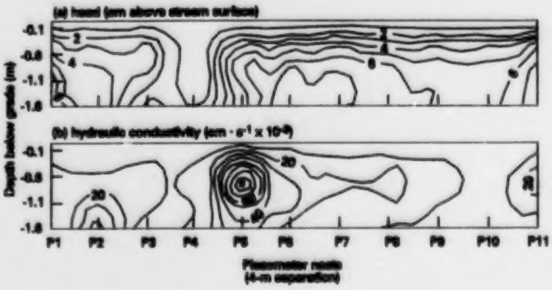


Fig. 6. Hydraulic head above the stream surface (*a*) and hydraulic conductivities (*b*) along the shoreline at Papineau Creek.



zone. Saturated sand 1.7–3.3 m thick dominated (1136 $m \cdot s^{-1}$) this area with possible gravel or cobbles (1395 $m \cdot s^{-1}$) in the vicinity of P2. Distinct pathways of flow from the terrestrial to the aquatic portion of the catchment were not apparent.

The mean discharge of groundwater in the reproductive habitats was 1.6×10^{-3} $cm \cdot s^{-1}$, or 1.2×10^{7} L of groundwater discharging annually.

Dickson Lake

The hydraulic heads in the nearshore zone at Dickson Lake indicated groundwater was discharging to the surface waters offshore (Fig. 5*a*). Flow converged from the north and south towards P4 and P7 along the shoreline. Offshore, there was a convergence at P3 and P4 with no flow apparent at P8. The hydraulic conductivities along the shoreline were greatest below 60 cm from P3–P5 and P7 (Fig. 5*b*). Conductivities increased offshore particularly at the substrate surface between P2 and P5. Conductivities also increased in the shallow substrate of P7 and P8. A pathway of flow approximately 14 m wide was suggested at P3–P5.

The thickness of the overburden material was estimated to be >7 m (no bedrock arrivals from the seismic survey were recorded). The unsaturated zone 2.3–3.2 m thick and composed of heterogeneously distributed unconsolidated sand, gravel, cobbles, and boulders (270–335 $m \cdot s^{-1}$). The saturated zone was composed of sand, gravel, and cobbles (1530–2696 $m \cdot s^{-1}$). The coarsest materials were located in the 30 m adjacent to the shoreline in the southwest (S1–S3).

The mean discharge of groundwater in the reproductive habitats was 1.7×10^{-3} $cm \cdot s^{-1}$, or 4.1×10^{7} L of groundwater discharging annually.

Papineau Creek

Hydraulic heads suggested an upwelling of groundwater towards the surface water dispersing from areas at P2 and P6–P10 (Fig. 6). Hydraulic conductivities suggested a primary lens of coarser materials 16 m wide and between 25 and 175 cm deep extending from P5 to P9 (Fig. 6). A second lens of coarser materials was apparent >100 cm deep at P2.

The thickness of the overburden material was estimated to be >10 m (no bedrock arrivals from the seismic survey were recorded). The unsaturated zone in the terrestrial catchment was 1.6–2.4 m thick and consisted primarily of sand (327–581 $m \cdot s^{-1}$). The saturated zone was composed of primarily sand (1532–1673 $m \cdot s^{-1}$). Coarser material (1737–1829 $m \cdot s^{-1}$) was located in the 20 m adjacent to the stream in the south (S1).

The mean discharge of groundwater in the reproductive habitats was 1.9×10^{-3} $cm \cdot s^{-1}$, or 2.6×10^{7} L of groundwater discharging annually.

Discussion

Successful brook trout reproduction in Canadian Shield waters requires coarse substrate materials <1 m thick with distinct and constant upwelling groundwater (Curry and Noakes 1995; Curry et al. 1995). Our observations indicate these habitats are manifestations of lenses of coarse materials that underlie the nearshore zone. The lenses can be restricted to the nearshore zone (Meach Lake), or extend at least 20 m into the terrestrial catchment (Dickson Lake and Papineau Creek). They displayed varied dimensions of >1 m thick and 10–16 m in width along the shoreline, sometimes with smaller, adjacent projections <5 m wide. Each lens directed groundwater flow into the spawning and incubation habitats at flow rates greater than adjacent nearshore areas (Curry and Noakes 1995).

Using the observed groundwater discharge and annual precipitation inputs, it is possible to estimate an area of recharge necessary for sustaining groundwater travelling through lenses and into the spawning and incubation habitats. Studies of hill-slope hydrology in similar catchments in humid climates indicate only a portion of precipitation recharges shallow, local groundwater with most lost to evapotranspiration and through quick-flow pathways (Devito et al. 1996; McDonnell and Taylor 1987). Using a conservative annual estimate of 100 cm precipitation and 40% recharge, the estimated area of recharge necessary to sustain the observed groundwater discharge is 3.0, 6.4, and 10.3 ha of the terrestrial subcatchment areas at Meach Lake, Papineau Creek, and Dickson Lake, respectively. Consequently, up to 97% (Meach Lake) or 10 ha (Dickson Lake) of the terrestrial catchment appears directly linked to brook trout spawning and incubation habitats.

Lenses are most likely buried glaciofluvial deposits (White 1974) such as eskers (possibly Dickson Lake), or remnants of glacial stream channels (Papineau Creek and Meach Lake). Reproductive habitats would be created when postglacial changes in surface water levels or erosion (glaciofluvial or present day channel cutting, e.g., Papineau Creek) exposed the lenses in the nearshore zone. This dynamic nature of groundwater development and pathways within glacial deposits indicates that the time when a lens could become available to brook trout and the duration of the lens's ability to support reproduction are variable and unknown. It is common in Canadian Shield waters for trout to maintain secondary spawning sites where there is significantly reduced activity or sporadic usage (R.A. Curry, unpublished data). Secondary sites may be new, developing, or older, deteriorating areas of discharging groundwater that represent the trout's evolutionary adaptation to the dynamic groundwater systems on which they depend.

A link between groundwater development and pathways in terrestrial catchments and brook trout reproductive habitats has critical implications for integrating sustainable forestry and fishery management. Increasing groundwater levels following removal of the forest cover (Peck and Williamson 1987) may enhance groundwater delivery to the nearshore zone and therefore the quality of spawning and incubation habitats. Alternatively, a shallower water table could alter the temperature of groundwater delivered to the nearshore habitats and affect incubation success (Hokanson et al. 1973). Successful incubation also requires a stable environment (Marten 1992). Groundwater temperature, chemistry, and flow are stable in spawning and incubation habitats (Curry et al. 1995). Stability may be jeopardized by fluctuating rates of groundwater discharge that occur after timber harvesting (Wright et al. 1990).

Preservation of brook trout spawning and incubation habitats during timber harvesting typically focuses on the direct protection of the nearshore zone. For example, a buffer zone or reserve adjacent to the shoreline can be created, e.g., ≤90 m (Ontario Ministry of Natural Resources 1988). A 90-m buffer zone would protect <9, <23, and <55% of the recharge areas required to sustain the reproductive habitats at Dickson Lake, Meach Lake, and Papineau Creek, respectively, assuming the critical areas for recharge are located adjacent to the nearshore zone. Such buffer zones may be inadequate if they do not encompass and protect the entire recharge zone associated with the discharge zone being used by trout. It would be more realistic to view the spawning and incubation habitats in terms of necessary hydrological requirements and thus develop a definition of a buffer zone in terms of hydrological units.

Other development activities within a catchment such as creation of reservoirs, removal of aggregate materials, and the construction of roads and human residences may alter groundwater recharge and impact spawning and incubation habitats. Development can also affect groundwater quality. Curry et al. (1993) observed road salt contamination of the groundwater in brook trout spawning and incubation habitats. Nutrients can also be introduced to groundwater discharging in the nearshore zone from human septic systems (Lee 1972) and agriculture (Peterjohn and Correll 1984). Protection from these activities will also require the recognition of the hydrological characteristics of spawning and incubation habitats.

Our evidence indicates a linkage exists between brook trout reproduction and catchment hydrogeology. There are clearly questions that remain unanswered regarding the specific hydrological relationships, as well as the impacts of land use on groundwater and brook trout reproduction. Nonetheless, present-day protection of habitats does not incorporate the hydrogeological characteristics of the spawning and incubation habitats. Protection guidelines must be reviewed to achieve the successful integration of terrestrial and aquatic resource management in watersheds inhabited by brook trout.

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The Kennedy Watershed Restoration Project: Identification of Forest Harvest Impacts and Opportunities for Salmon Stock and Habitat Rehabilitation in Clayoquot Sound, British Columbia

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Abstract

The Kennedy Watershed Restoration Project (KWRP) is currently organized into five separate studies designed to address two general working hypotheses. The first hypothesis is that forest harvest practices tend to fragment and simplify habitats, disrupt salmon life cycles, and precipitate salmon population declines. The second is that specific habitat rehabilitation prescriptions may be used to accelerate the recovery of depressed salmon stocks and associated ecosystem functions where damage associated with past forest harvest practices has occurred. The five individual KWRP projects include: (1) Retrospective analysis of salmon production variations, (2) Synoptic surveys of adult and juvenile salmon abundance and habitat use, (3) Synoptic surveys of habitat conditions, (4) Habitat restoration projects, and (5) the Kennedy Watershed habitat and salmon atlas. The KWRP studies have been designed to: (a) improve our understanding of forest harvest impacts on salmon populations and habitats, (b) support salmon stock rebuilding and habitat rehabilitation in the Kennedy Watershed of Clayoquot Sound, (c) test approaches used to rehabilitate habitats and rebuild stocks in order to identify critical production bottlenecks for salmon, and (d) develop expert system tools and apply an adaptive management approach to facilitate involvement of a diversity of stakeholders from industry, government, and community groups in resolving land use and resource management issues that frequently arise in British Columbia Watersheds.

Johannes, M.R.S. and Hyatt, K.D. 1998. The Kennedy Watershed Restoration Project: identification of forest harvest impacts and opportunities for salmon stock and habitat rehabilitation in Clayoquot Sound, British Columbia. Pages 389–401 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

Clayoquot Sound, on the west coast of Vancouver Island British Columbia (Fig. 1), has become an international icon in the debate about the impacts of forest harvesting practices on some of the last intact old growth forest ecosystems of coastal North America. Stimulated in part by international debate, the government of British Columbia has adopted a new province-wide Forest Practices Code (British Columbia Ministry of Forests and British Columbia Ministry of Environment 1995) to promote sustainable commercial and recreational use of timber, fiber, fish, wildlife and land resources. In addition, a land-use decision process has been developed for specific application in Clayoquot Sound to promote sustainable forest resource extraction practices that will address concerns about the maintenance of ecosystem integrity from the perspective of both aboriginal and non-aboriginal community values. New initiatives and regulations, resulting from the Clayoquot Sound Scientific Panel recommendations, have imposed unprecedented change on forest management in Clayoquot Sound as part of a new, adaptive management approach to sustainable resource use there (Scientific Panel for Sustainable Forest Practices in Clayoquot Sound 1995a, b).

In 1994, the not-for-profit Northwest Ecosystem Institute (NEI) in collaboration with participants from the Tla-o-qui-aht First Nation, forest industry (e.g., MacMillan Bloedel Ltd.), public interest groups (e.g., Thornton Creek Salmon Enhancement Society, Clayoquot Biosphere Project, Interrain Pacific, Long Beach Model Forest Society) and government agencies (e.g., British Columbia Ministry of Forests, British Columbia. Ministry of Environment, Lands and Parks, Canadian Department of Fisheries and Oceans) initiated an interdisciplinary habitat and salmon stock restoration program on the Kennedy Watershed. The general purpose of the program was to investigate the impacts of more than 60 years of forest harvesting on salmonid habitats and populations in order to identify and then develop options to implement stock rebuilding and habitat restoration where damage could be identified.

A diversity of factors supported the choice of the Kennedy Watershed as the focus for this program. First, at 54 700 ha, the Kennedy Watershed is one of the largest watersheds within Clayoquot Sound. Second, the watershed has historically accounted for some of the highest forest and fisheries resource values in the region (Brown et al. 1987, The Scientific Panel for Sustainable Forest Practices in Clayoquot Sound 1995b), and its natural resources are considered to be one of the keystones to the socioeconomic well being of both aboriginal and non-aboriginal peoples from the nearby communities of Ucluelet, Tofino, and Opitsat, among others. Third, several sub-basins within the watershed (e.g., Upper Kennedy, Sand River, and Muriel Lake) have supported extensive forest harvesting operations, which were initiated more than 60 years ago when roads were first built into the region (Johannes and Hyatt 1996, Guppy 1997). The result is that a higher proportion of this watershed has experienced land-use disturbances over a longer interval than most comparable areas along the west coast of Vancouver Island (Brown et al. 1987; Carruthers et al. 1997). Thus, the watershed presents ample opportunities to identify signature impacts of forest resource extraction activities (e.g., road building, clearcut harvest, log transport, log storage) on terrestrial and aquatic habitats along with the fish and wildlife populations that depend on them. Fourth, by contrast, several sub-basins of the Clayoquot Valley portion of the watershed have remained virtually free of the impacts of land-based, resource extraction activities. Therefore, they offer excellent prospects for assembling unharvested-area, control observations that are essential to help differentiate resource responses that are associated with factors other than forest harvesting activities (e.g., climate change and fisheries exploitation). Finally, the Kennedy Watershed offers superior opportunities to retrieve or develop reliable inventory information on both the current and long-term state (i.e., 1900 to present) of key resources such as forest cover, aquatic habitat conditions, and temporal changes in populations of local salmonid species. This is due to the exceptional diversity of observations and information available from: (i) knowledgeable, long-term residents of nearby aboriginal and non-aboriginal communities, (ii) historic records of resource extraction activities held in forest industry archives, (iii) historic assessment programs conducted on forest and salmonid population resources within the watershed by government agencies (e.g., Canadian Department of Fisheries and Oceans, British Columbia Ministry of Forests and British Columbia. Ministry of Environment Lands and Parks) and (iv) recent assessment programs conducted by both government agencies and public interest groups (e.g., Tla-o-qui-aht First Nation, Northwest Ecosystem Institute, Nuu-chah-nulth Tribal Council, Clayoquot Biosphere Program, Long Beach Model Forest, Thornton Creek Salmon Enhancement Society) on a variety of fish and forest resources.

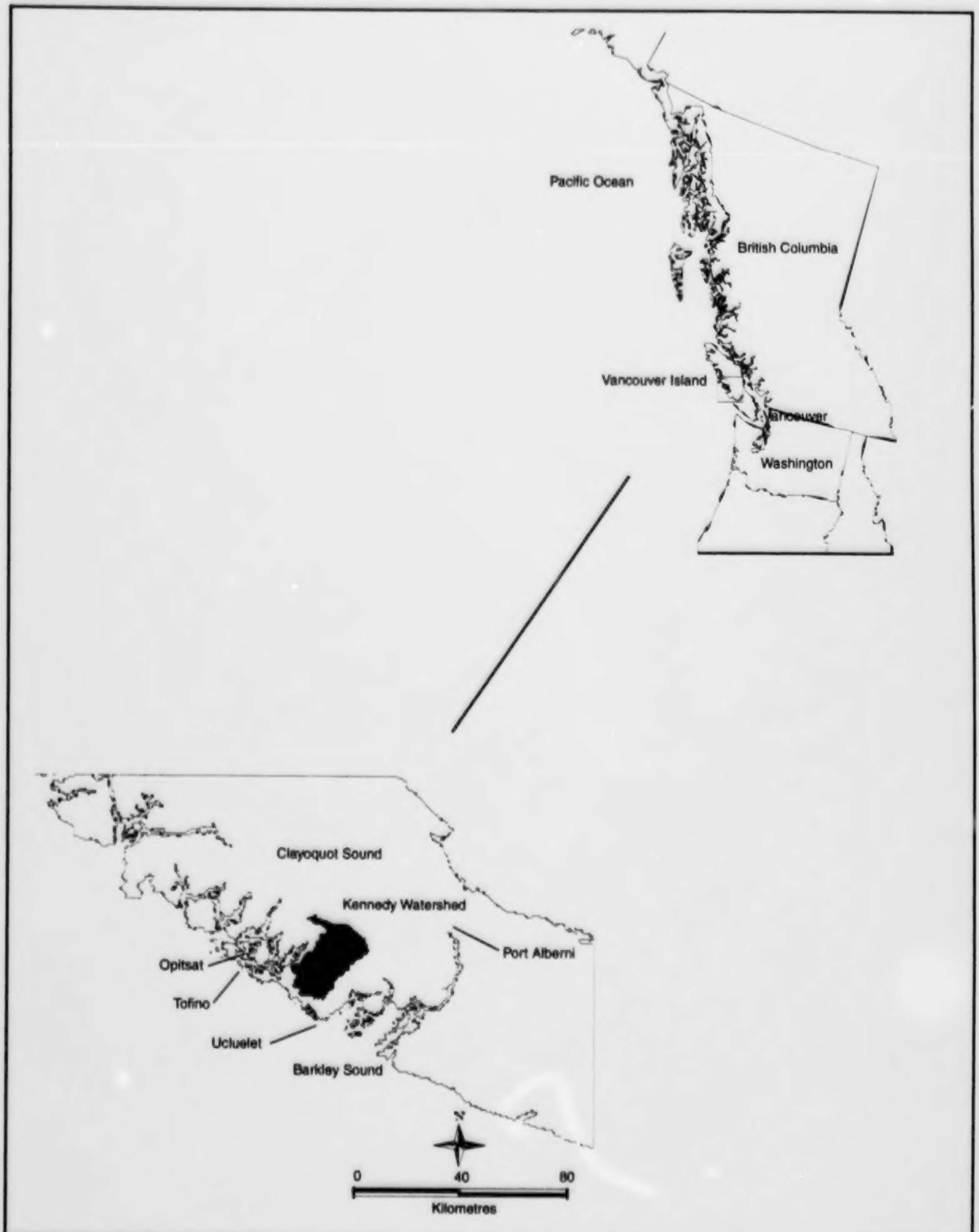

Figure 1. Location of the Kennedy Lake Watershed in Clayoquot Sound, Vancouver Island, British Columbia.

The Kennedy Watershed Restoration Project (KWRP), described in the article that follows, has been developed with input from local stakeholder groups concerned about rebuilding all stocks of salmon and especially the sockeye populations originating in the Kennedy Lake Watershed (Gill et al. 1997). Component studies of KWRP described below currently receive funding or support from Forest Renewal British Columbia., the B. C. Science Council, Canadian Department of Fisheries and Oceans, B. C. Ministry of Environment, Lands and Parks and the British Columbia Ministry of Forests. In the remainder of this paper we will provide an overview of existing background information on salmon stock and habitat status in the watershed, outline the strategic approach adopted to guide future KWRP activities, identify current objectives, and briefly review project progress to date in meeting these objectives.

Watershed Background and History

The Kennedy Watershed is located within Clayoquot Sound on the west coast of Vancouver Island, within 5 km of Tofino or Ucluelet, 25 km of Opitsat and 80 km of Port Alberni (Fig. 1). The Kennedy Watershed drains an area of 54 700 ha in a glacial, U-shaped valley surrounded by mountains. Vegetation comprises a coastal temperate rainforest with annual precipitation of more than 700 cm, principally between mid-October and May. Extreme weather events are not uncommon, creating particularly intense rainfall over short periods lasting from hours to days (Redmond and Taylor 1997). The combination of high precipitation and highly erodible glacial deposits (sand and gravel) on steep terrain produces considerable geological instability in the area.

Approximately 60% of the watershed area supports harvestable timber divided among three Tree Farm Licences (TFLs) including MacMillan Bloedel Ltd., International Forest Products, and Crown Reverted Lands. The remaining 40% of the watershed comprises Pacific Rim National Park, Kennedy Lake Provincial Park, protected Crown lands, highway and hydro corridors, Tla-o-qui-aht First Nation reserves and lakes (e.g., Kennedy, Muriel, Clayoquot, Angora). Forest harvesting, commercial fishing, mining, recreation and tourism are the main socioeconomic activities supported by natural resources in the area.

Removal of forest cover by logging activities has occurred over 12% of the total area within the watershed (Johannes and Hyatt 1996). However, concentration of these activities in more readily accessible areas (e.g., Upper Kennedy River, Sand River and Angora Ridge, Muriel Lake, Kennedy Flats) has resulted in losses of the original forest cover from more than 60% of the riparian zone forests along 300 km of riverine and stream habitats known to be used by several species of salmonids as spawning and nursery areas. Forestry activities such as road construction and clearcuts in combination with terrain and climate conditions have created potentially high sediment erosion along with frequent debris and landslide events in many high elevation subbasins of the watershed. The latter processes, in combination with alterations associated with changes to riparian zone canopy cover and altered recruitment of supplies of large woody debris, have considerable potential to have influenced habitat conditions and production trends for salmonids throughout a significant portion of the watershed over the past several decades.

Anadromous salmon occupy at least 60 subbasins within the Kennedy Watershed and its habitat, and salmon production values are considered among the highest found anywhere within Clayoquot Sound (Brown et al. 1987; Hyatt 1998). Anadromous salmon and resident salmonids that use the Kennedy Watershed include sockeye (*Oncorhynchus nerka*), coho (*O. kisutch*), chinook (*O. tshawytscha*), steelhead (*O. mykiss*), chum (*O. keta*), pink salmon (*O. gorbsucha*), rainbow trout (*O. mykiss gairdneri*), cutthroat trout (*O. clarki clarki*) and Dolly Varden char (*Salvelinus malma*). Given the size and complexity of the watershed, there are dozens of identifiable local populations of salmon and trout, exhibiting a diversity of life history traits (Johannes and Hyatt 1996; Hyatt unpublished data).

At the time of first contact with non-aboriginal peoples, the wealth of marine resources, including salmon, in the Clayoquot Sound area supported some of the highest population densities of indigenous peoples found anywhere in the Americas (Arima 1983). Details concerning seasonal patterns of exploitation, methods of harvest and importance of salmon to both diet and culture may be found in Drucker (1951), Arima (1983), Bouchard and Kennedy (1990); however, it is sufficient to note here that salmon represent an irreplaceable link to the past and an invaluable resource for both present and future members of First Nations communities of the area. Similarly, stocks of anadromous salmon originating from the Kennedy Watershed have played a key role in supporting the socioeconomic activities of both aboriginal and non-aboriginal communities proximal to the watershed from the prehistoric era to

present (Guppy 1997). However, throughout the past century stocks of many species of salmon in Clayoquot Sound and the Kennedy Watershed appear to have declined in parallel with increases in resource extraction (e.g., fisheries exploitation, forest harvest, mining) and associated changes in land use activities (road building, urbanization) known to have either direct or indirect effects on salmon stocks and their habitats (Johannes et al. 1997). Thus, local communities have been witness to local fisheries resource declines that appear to be an outcome of a failure on the part of resource managers. The latter group have repeatedly failed to plan and control resource extraction activities such that crucial ecosystem linkages are maintained and multiple resource values are sustained at scales involving individual watersheds and local populations of salmon. Both the new British Columbia. Forest Practices Code, that applies to all Crown Lands in the province, and the Scientific Panel recommendations, that apply only to Clayoquot Sound, emphasize the need for a change in resource management focus. This entails movement away from maximization of harvestable biomass of one or a few species over extensive and often arbitrary geographic areas, towards optimization of sustainable harvest of many species within a functional ecosystem and specific watershed context (British Columbia Ministry of Forests and British Columbia Ministry of Environment 1995, Scientific Panel for Sustainable Forest Practices in Clayoquot Sound 1995b). The need to develop a science of sustainable fisheries and forest management at the watershed and local stock level is the motivating force behind development of an interdisciplinary Kennedy Watershed Restoration Program (KWRP).

KWRP Program Objectives and Study Design

The Kennedy Watershed Restoration Program has been designed to address two general hypotheses. The first is that forest harvest practices tend to fragment and simplify habitats, disrupt salmon life cycles and precipitate salmon population declines. The second is that specific habitat rehabilitation prescriptions may be used to accelerate the recovery of depressed salmon stocks and associated ecosystem functions where damage associated with past forest harvest practices has occurred (Forest Renewal B.C. 1995). Thus, the general strategy for the KWRP entails integration of results from retrospective analyses and synoptic studies of current fisheries and forest resource states with experimental manipulations consisting of specific habitat rehabilitation projects such that we may 1) identify the impacts of historic logging practices on salmon populations and habitats through retrospective analyses of their condition before, during, and after logging events; 2) initiate synoptic surveys on the current state of salmon populations and habitats affected by ongoing forest harvesting; and 3) identify, design, and then apply specific stock and habitat rehabilitation initiatives to test for critical associations between logging-induced habitat changes and salmon production limitations.

The KWRP activities are currently organized into five component studies to achieve the objectives above. These studies are discussed below.

Retrospective Analysis of Salmon Production Variations

This study, led by personnel from the Department of Fisheries and Oceans (contact K. Hyatt), is meant to examine long-term (> 25 years) changes in productivity of adult salmon populations in relation to temporal variations in both natural and anthropogenic factors operating in local freshwater and more distant marine environments. Time series analysis and comparisons for similarities or differences in production trends exhibited by salmon populations originating from the Kennedy Watershed, as opposed to other nearby watersheds that have a different history of resource interactions (e.g., Barkley Sound, Hyatt and Steer 1987), will be useful in testing for the involvement of forest harvest impacts versus other factors (e.g., marine climate and fisheries exploitation patterns) in controlling long-term production trends of salmon.

Current Status of Salmonids

Personnel from the Northwest Ecosystem Institute (contact M. Johannes) coordinated synoptic surveys between 1994 and 1997 to determine the current distribution and abundance patterns of juvenile and adult salmonids in several logged and unlogged sub-basins of the Kennedy Watershed (e.g., Johannes and Hyatt 1996). Participants in survey efforts originated from First Nations groups (Tla-o-qui-aht First Nation fisheries personnel), public interest groups (Thornton Creek Salmon Enhancement Society, Clayoquot Biosphere Project), private industry (MacMillan Bloedel, International Forest Product, M.C. Wright and Associates), and government agencies (Fisheries and Oceans Canada). Comparisons of current population state versus historic levels identified in retrospective studies will permit determinations of whether specific species or populations within species are currently healthy or depressed.

Current Status of Salmonid Habitats

Personnel from the Northwest Ecosystem Institute coordinated the execution of condition assessment surveys according to Forest Renewal British Columbia. guidelines between 1994 and 1997 to determine the current state of stream and riparian habitat features of importance to salmonids in several logged and unlogged sub-basins of the Kennedy Watershed (Johannes et al. 1997). Results from this project are combined with those on the current state of various life history stages of salmonid populations to identify likely associations between salmon population state and forest harvest impacts on salmonid habitats.

Habitat Restoration Projects

In this project, results from assessments are used to identify and prioritize specific stock and habitat rehabilitation options to remedy past forest harvesting impacts on salmonids and their habitats. Design and testing of specific habitat restoration prescriptions is being conducted collaboratively among consulting biologists, aquatic ecologists, hydrologists, and engineers working through NEI with personnel from the B.C. Ministry of Environment Lands and Parks, B.C. Ministry of Forests, Forest Renewal B.C., and the Habitat and Enhancement Branch of the Canadian Department of Fisheries and Oceans. In our view, habitat rehabilitation projects in the watershed are likely to serve as valuable experimental tools to critically test for hypothesized associations between forest harvest impacts and salmon production limits.

Kennedy Watershed, Habitat and Salmon Atlas (KWAT)

Observations on the status of physical conditions and biological resources within watersheds are commonly gathered at such a diversity of spatial and temporal scales by so many groups (e.g., government agencies, private industry, public interest groups) that storage and organization of the information for effective retrieval at a later date has generally been restricted to mixtures of scattered paper and electronic archives with few, if any, existing relational links. Consequently, resource managers, planners, and a host of community groups experience great difficulty in retrieving well-integrated information on the state of forest, salmon, and wildlife resources or ecological processes that link them within watersheds. In the Kennedy Watershed, the quantity and quality of resource inventory information, along with the diversity of temporal and spatial scales on which it is being generated, provide a unique opportunity to create a spatially referenced atlas of landscape, forest, wildlife, salmon and habitat resources. Accordingly, the Northwest Ecosystem Institute, with funding from the British Columbia. Science Council and Forest Renewal British Columbia., and in partnership with personnel from the Canadian Department of Fisheries and Oceans and Interrain Pacific, is creating a multivolume atlas of landscape, forest, wildlife, salmon and habitat resources within the Kennedy Watershed.

The specific objectives of the KWAT project are to:

1) assemble, integrate, and document spatial and temporal data on climate, forest cover, salmonid populations, wildlife populations and their habitats in a multivolume atlas on the Kennedy Watershed;
2) facilitate the transfer of Kennedy Watershed information to resource managers, First Nations, industry and public interest group stakeholders;
3) develop a suite of ecological analysis and modeling tools that integrate both spatial and temporal resource data to facilitate joint involvement of community-based resource stakeholders, private industry, and government agency personnel in development of land-use planning and resource management decisions;
4) test the utility, to the various stakeholder groups, of the watershed-based information sets and tools developed above.

Study Progress and Results to Date

Retrospective Analysis of Salmon Production Variations

We are part way through a time series reconstruction of salmon abundance to support our retrospective analysis project goals. Over a century of historic observations on commercial catch of sockeye salmon, virtually all of which originate from Kennedy Watershed nursery areas, have been assembled and verified (Fig. 2). These observations confirm several inferences about historical trends in anadromous salmonid production originating from the Kennedy Watershed as follows. First, production of sockeye salmon has supported significant commercial fisheries within terminal areas of Clayoquot Sound for most of the past century. Second, throughout the period of record there have been numerous instances of apparent collapse (i.e., 1903, 1916, 1922, 1928, 1942–45, 1958) followed by rapid

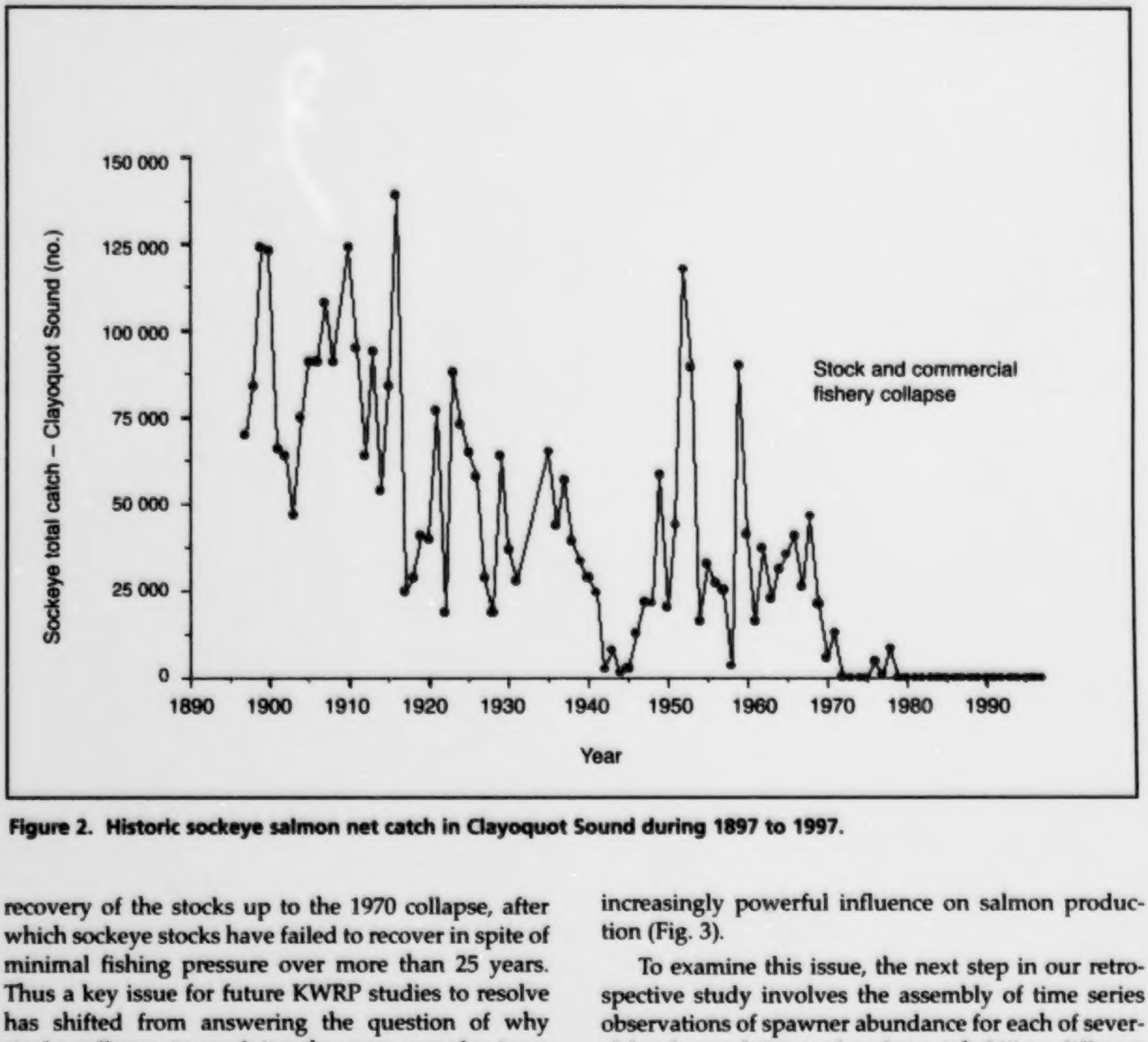

Figure 2. Historic sockeye salmon net catch in Clayoquot Sound during 1897 to 1997.

recovery of the stocks up to the 1970 collapse, after which sockeye stocks have failed to recover in spite of minimal fishing pressure over more than 25 years. Thus a key issue for future KWRP studies to resolve has shifted from answering the question of why stocks collapse, to resolving the more complex issue of why sockeye stocks, that have exhibited considerable resilience for recovery after repeated past collapses, should fail to recover from an apparently similar event initiated in the late 1960s. Although we have not fully explored this issue at this stage of KWRP, preliminary review of spatial and temporal elements of forest harvest and road building impacts support the notion that degradation of freshwater spawning or rearing areas of sockeye would have had minimal influence on their production prior to the decade of the 1960s. However, during later decades sockeye spawning and rearing areas, such as the Upper Kennedy River, Sand River, the Muriel subbasin and Cold Creek, were subjected to road building and logging activities that may have had an increasingly powerful influence on salmon production (Fig. 3).

To examine this issue, the next step in our retrospective study involves the assembly of time series observations of spawner abundance for each of several local populations of sockeye inhabiting different areas of the watershed that have either been heavily influenced or remain untouched by forest harvest activities. Thus, it appears that careful assembly and analysis of both historic and current sockeye stock observations offers some potential for teasing out some of the signature impacts of logging from the often confounding, simultaneous influences of fisheries exploitation, climate change, road building, and general human disturbance.

Analysis of information on other species of salmon also indicates a pattern of general decline. Sockeye continue to account for a similar percentage of the commercial catch of all salmon species in Clayoquot Sound both before and after the persistent collapse of Kennedy sockeye in the late 1960s (Fig. 4).

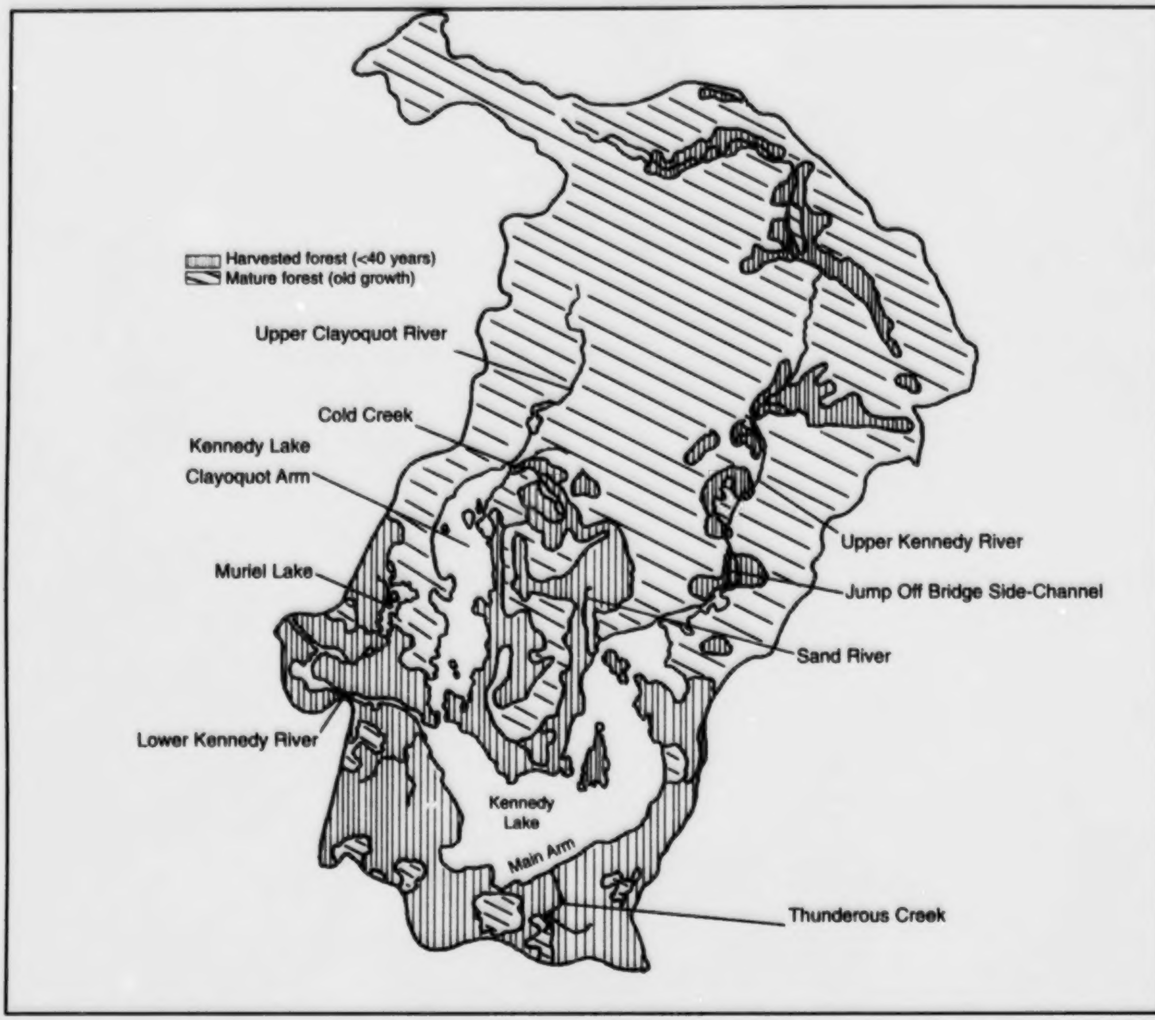

Figure 3. Extent of forest harvesting activities to 1990 across the Kennedy Lake Watershed.

This result suggests either that Kennedy sockeye were replaced with catches from other sockeye stocks migrating through waters off Clayoquot Sound or that other species of salmon suffered declines in production similar to those of Kennedy sockeye. Time series observations on annual production variations for anadromous salmon originating within the Kennedy Watershed are unavailable for species other than sockeye because they are caught in mixed-stock fisheries in marine waters. Accordingly, retrospective analyses involving other salmon species are unlikely to be as rewarding with respect to identifying clear effects of forest harvest activities on long-term production trends. These analyses will need to include within watershed observations of distribution and abundance of stocks and habitat condition to serve as our main sources of information regarding links between forest harvest practices and production trends.

Current Status of Salmonids and their Habitats

Results from synoptic surveys of adult and juvenile salmonid abundance and habitat use in test (i.e., logged) and control (i.e., unlogged) areas throughout the Kennedy Watershed were gathered under both high and low seasonal flow conditions during 1994–1997. They support the general expectation that salmon in logged streams would exhibit evidence of

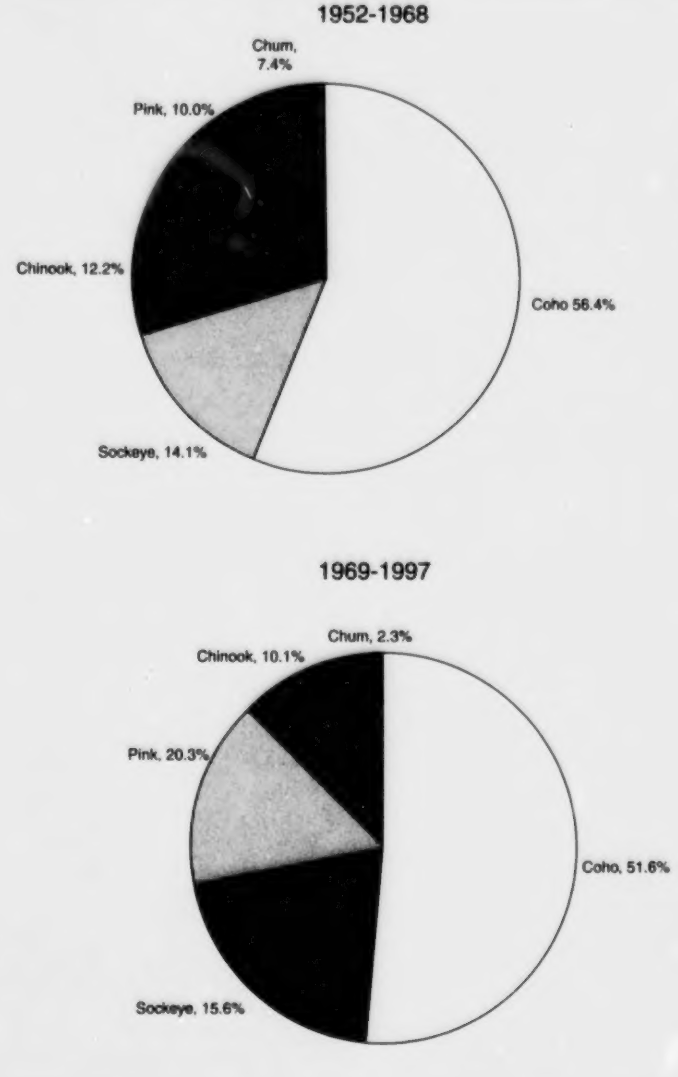

Figure 4. Percentage contribution of individual salmon species to total commercial fishery catch in Clayoquot Sound during pre sockeye stock collapse (1952 to 1968), and during post sockeye stock collapse (1969 to 1997).

lower stock productivities associated with identifiable disruptions in patterns of habitat use for spawning, incubation and rearing. Comparisons between large, third order streams including Upper Kennedy River and Upper Clayoquot River, and small, first order streams including Muriel S1 Creek and Cheewhat S2 Creek (an unlogged control stream of similar order located outside of the Kennedy Watershed) were used to identify changes in the quality and quantity of salmon habitat associated with logged and unlogged forest states (Fig. 5).

Logged streams exhibited evidence for changes in the distribution, abundance, and state of habitats that are essential to sustained production of salmonids in freshwater ecosystems. Changes to patterns of recruitment and loss of substrate, large woody debris, and pools appear responsible for most of the logged versus unlogged habitat differences in

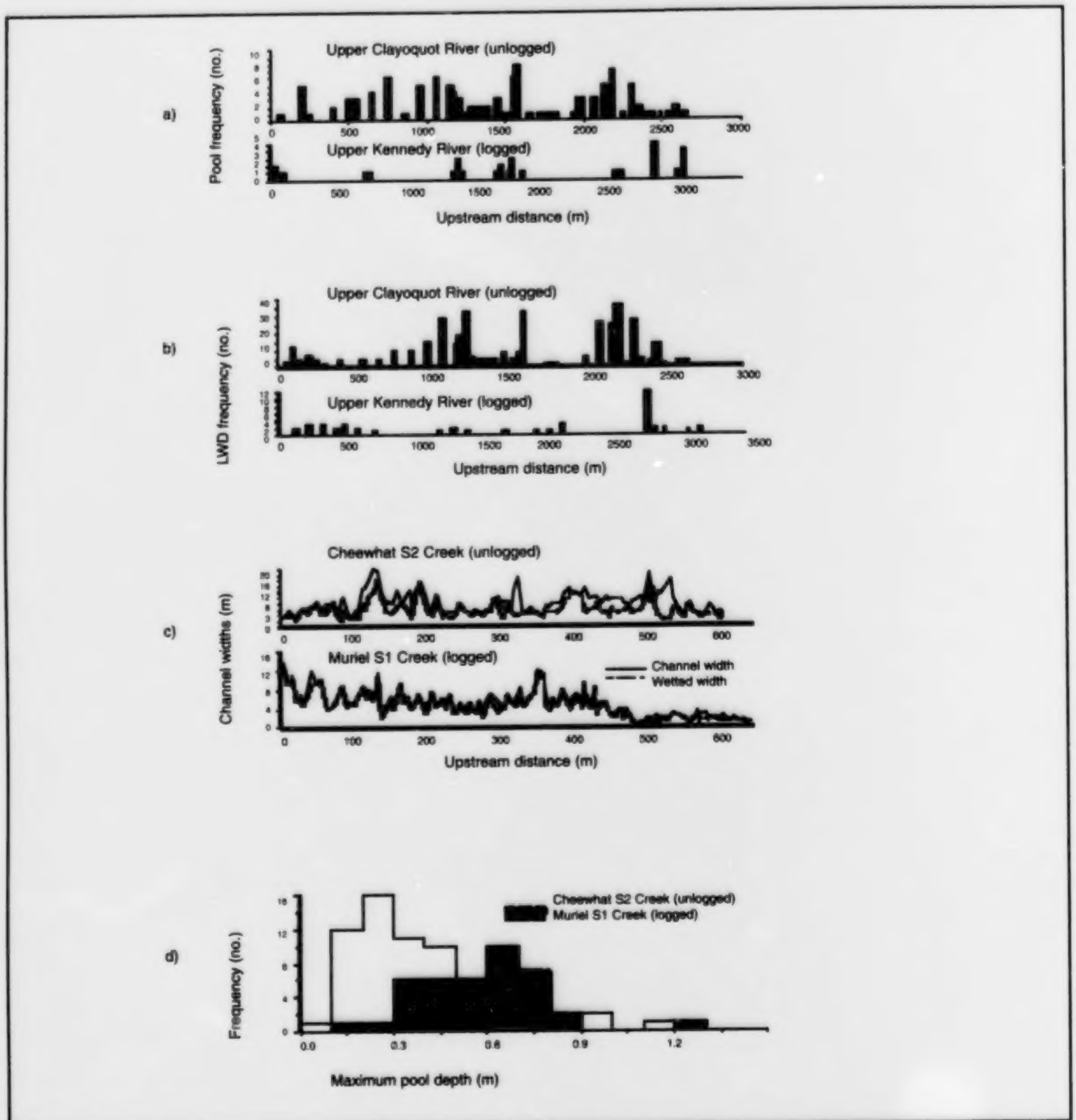

Figure 5. Habitat conditions in logged and unlogged large and small order streams showing comparisons between: (a) pool frequency in the Upper Clayoquot and Kennedy Rivers, (b) large woody debris frequency in the Upper Clayoquot and Kennedy Rivers, (c) channel widths in Muriel S1 and Cheewhat S2 Creeks, and (d) depth distribution of pools in Muriel S1 and Cheewhat S2 Creeks.

both large (Fig. 5a, b) and small order (Fig. 5c, d) streams. Patterns of salmonid habitat use differed greatly in logged and unlogged streams. Juveniles and adults exhibited more equitable use of off-channel and mainstem habitats in unlogged streams. Spawners were often confined to the lower, mainstem portions of small logged streams, and salmon production (as indexed by mean size and abundance of rearing juveniles or total abundance of adult spawners) was often lower in logged streams.

Although local stock productivity indicators such as numbers of spawning adults or rearing juveniles exhibited high annual variability, some general trends appear to be emerging from comparisons of abundance trends in logged versus unlogged areas. For example, sockeye spawning and nursery areas in the unlogged Clayoquot sub-basin support a higher mean abundance of both rearing juveniles and spawning adults than the heavily logged Upper Kennedy River and Main Arm basin, even though the latter possess larger useable areas for sockeye spawning and rearing than the Clayoquot sub-basin does. Similarly, comparisons of adult and juvenile production indices for both sockeye and coho salmon in the small but heavily logged Muriel sub-basin relative to observations of these same species in the undisturbed Cheewhat Watershed (carefully selected as a matched control area) indicate that mean annual production of both species is approximately 5–10 times greater for the unlogged system.

We do not regard the observations and preliminary analyses undertaken to date to be unequivocal evidence for logging as the major determinant of salmonid production trends throughout the Kennedy Watershed. However, we do stress that differences in the condition of habitats and production of local salmon stocks associated with logged versus unlogged systems are consistent with the notion that past forest harvest practices have degraded salmon habitats and depressed their populations. Because the mechanisms that mediate these changes are still poorly understood, additional field assessments and analytical work are required to sort out what are clearly complex interactions between watershed processes and resource extraction practices. However, in some instances we have concluded there is sufficient evidence for strong enough links among past forest harvest practices, habitat change and local stock declines to identify, design, and then begin implementation of high-priority habitat and salmon stock rehabilitation projects in specific locations within the Kennedy Watershed.

Habitat Restoration Projects

Quantifying and understanding the entire suite of land use impacts on habitats and then developing prescriptions to reverse them, such that local salmon stocks may be rebuilt, is still considered to be a highly experimental activity likely to be accompanied by uncertain outcomes (Hartman and Miles 1995). Although we agree with this assessment, we also suggest that one of the most effective means of obtaining knowledge about truly critical links between land-use practices, habitat change, and salmon population variations will be through the use of carefully designed restoration projects. These projects should be considered not only for their future potential production benefits but also for the experimental information yield they can provide to improve our understanding of associations among land-use practices, habitat state, and fish production variations.

To address degraded habitat conditions, fragmented habitat use, and associated salmon stock production declines, we have identified, designed, and initiated stream restoration projects. Projects are designed to assist in the rebuilding of selected salmon stocks where the case for historic to recent logging impacts appears strong and the future benefits relative to cost appear high. During the past 2 years, two rehabilitation projects have reached the implementation stage in the Kennedy Watershed while another 10 projects are in the planning and pre-implementation review stage.

The most significant rehabilitation project initiated thus far is the Jump-Off Bridge side-channel project on the Upper Kennedy River (Fig. 3). Local populations of salmon originating from the Upper Kennedy River have dropped precipitously in the last four decades in a pattern that suggests a link to logging and road building activities in several subbasins in the area. Comparison of patterns of salmon recruitment and current state of habitat conditions between heavily logged Upper Kennedy River and unlogged Clayoquot River suggest that extensive loss of side channel habitat, formerly used by spawning salmon, appeared in part to be responsible for the salmon production bottleneck. Historic time series surveys of annual variations in spawner abundance and subsequent recruitment of fry into the main arm of Kennedy Lake by the Canadian Department of Fisheries and Oceans, suggested that a production bottleneck had been created in the egg incubation environment. The KWRP results also suggested a probable cause of limited egg incubation environments related to changing patterns in sediment and

debris loading associated with historic logging activities in the Upper Kennedy River sub-basins. Synoptic surveys of habitat conditions identified a dewatered sidechannel in the Upper Kennedy River as an attractive rehabilitation opportunity (Fig. 3).

The Jump Off Bridge side-channel project was initiated to begin the long process of restoring salmon habitat in the Upper Kennedy River to a prelogging state and to assist in rebuilding local populations of sockeye and coho salmon. Design and construction plans were developed specifically to (1) provide stable spawning habitat for salmonids; (2) ensure continuous access to the channel for migrating salmon during low flows; (3) improve access and extent of existing juvenile salmon rearing habitat, and (4) prevent further decline of existing habitat by limiting excessive recruitment of substrate and woody debris from the mainstem river into the side channel.

Excavation and construction involved the upper 1250 m of the side channel so as to provide stable flow throughout the remaining 1200 m of relic channel. Channel habitat complexes (riffles, pools, and glides) were designed to seasonally accommodate adult salmon holding, adult salmon spawning, and juvenile salmon rearing. The project began in mid-fall, 1996, with the construction of flood protection dykes and was virtually completed with the opening of the water supply in late summer 1997. The channel presently functions on 0.40 m^3/sec average discharge. More than 150 pairs of sockeye and coho salmon have been observed spawning in the channel during 1997/98. The site will be monitored for at least 5 years for adult salmon spawner abundance, habitat use and associated juvenile recruitment, to both riverine and lacustrine habitats.

Kennedy Watershed, Habitat and Salmon Atlas

The NEI and its partners have begun to assemble and document the sources of historic to recent resource inventory data for inclusion in a multivolume Kennedy Watershed Resource Atlas. Both temporal and geographic information system databases have been developed to keep track of the many datasets acquired to include in various volumes of the atlas. Although atlas volumes will eventually be produced to consider resource inventory observations at several different spatial and temporal scales, our initial focus has been to assemble observational data pertaining to scales involving the whole watershed (e.g., Fig. 3) over the full temporal period of record. Thus, a draft of Volume 1 of the atlas is nearing completion and its contents include a general introduction followed by chapters dealing with: (i) aquatic resources and the physical environment, (ii) forest resources, (iii) salmon resources, (iv) fish, forest and aquatic ecosystem dynamics and (v) disturbance and cumulative impact effects of land use practices on forest habitats and the salmonids that use them. This draft volume will be distributed to representatives of community, industry and government stakeholder groups in the near future for their comments and input. Subsequent volumes of the atlas to be produced over the next 2–3 years will consider development of information on resources and ecosystem processes at finer levels of spatial and temporal resolution, along with the development of analytical tools and models. The ultimate objective of the KWAT project is to integrate the contents of the resource inventory databases, multivolume atlas, and analytical tools or models into a prototype for a watershed-based, expert, information system. This may be used to address questions and decisions about land-use/resource management issues that are frequently of common interest to a diversity of stakeholders from private industry, government agency, and public interest groups.

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Overwinter Sedimentation of Clean Gravels in Simulated Redds in the Upper Grande Ronde River and Nearby Streams in Northeastern Oregon, USA: Implications for the Survival of Threatened Spring Chinook Salmon

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Abstract

We monitored sedimentation of fine sediment (diameters < 6.3 mm) in cleaned gravels in artificially constructed redds during the incubation season for spring chinook salmon as well as the fraction of the streambed covered by fine sediment (%) in streams with different levels of surface fine sediment (%). Fine-sediment accumulation was highly variable, but occurred consistently, indicating that fine sediment is transported invariably during the winter. The magnitude of sedimentation was related to surface fine sediment in a statistically significant fashion when data from all streams in all years were analyzed ($p < 0.01$); this was not the case in a single year among streams nor in the upper Grande Ronde River among all sampling years. Sedimentation was the highest in the upper Grande Ronde River where surface fine-sediment levels were highest. We conclude that the winnowing of fine sediment from redds by salmon is a transient condition in the monitored streams, especially where surface sediment is high. The magnitude of overwinter sedimentation collected in containers in constructed redds in the upper Grande Ronde River was not related, in a statistically significant fashion, to stream discharge. In the upper Grande Ronde River, it appears that the magnitude of sedimentation during the incubation period is not limited by the discharge or the availability of mobile fine sediment because surface fine-sediment levels are high and stream discharge regularly occurs at magnitudes that are adequate to transport fine sediment. It appears that overwinter sedimentation is reducing salmon survival-to-emergence in the study area and especially in the upper Grande Ronde River. Surface fine sediment appears to provide a statistically significant index of the susceptibility of redds to overwinter sedimentation in streams.

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Introduction

Populations of spring chinook salmon (*Oncorhynchus tshawytscha*) in the Grande Ronde River in Oregon, U.S.A., have declined precipitously since late 1960s (Anderson et al. 1993); only three spring chinook salmon redds were found in the Upper Grande Ronde in 1994. These populations of salmon have been listed as threatened under the Endangered Species Act. While high levels of mortality at downstream hydroelectric facilities are a major cause of the declines, habitat degradation has also contributed significantly by reducing egg-to-smolt survival (Anderson et al. 1993; National Research Council 1996). The river has high levels of fine sediment caused by elevated sediment delivery from roads, logging, mining, and grazing, and natural sources, including fire (Anderson et al., 1993). During the past 50 years, the majority of large pools in the river have been lost (McIntosh et al. 1994); elevated sediment delivery from land management has probably contributed to pool loss (McIntosh et al. 1994). Surface fine sediment in spawning habitat in the upper Grande Ronde River ranges from about 30 to 50%; these levels of fine sediment have been found to result in low salmon survival from egg to parr and low densities of rearing salmon in streams similar to those in our study area (Scully and Petrosky 1991).

The effects of fine sediment on salmon survival-to-emergence is a critical biological issue with major ramifications for watershed management because of the depressed status of chinook salmon populations in the Grande Ronde River. Reduced survival caused by fine sediment is a source of density-independent mortality that can reduce salmon numbers even at a low population density (Everest et al. 1987), such as occurs in the Grande Ronde. If salmon survival-to-emergence in the Grande Ronde River is to be increased, fine sediment levels must be reduced. A reduction in sediment will require curtailing land management activities that cause fine sediment to be deposited in spawning habitat.

Field and laboratory studies indicate that as fine sediment levels in spawning substrate increase, salmon survival from egg to fry is reduced by multiple effects including entombment and reduced flow of oxygenated water to the eggs (Everest et al. 1987; Chapman 1988). It has been posited that high levels of fine sediment in ambient stream substrate may not reduce egg-to-fry survival in field conditions because salmon winnow fine sediment from the redd during the act of spawning (Everest et al. 1987; Chapman 1988). However, it has been documented in laboratory and field settings that sedimentation of fine sediment occurs in cleaned gravels during sediment transport and in salmonid redds subsequent to spawning (Meehan and Swantson 1977; Beschta and Jackson 1979; Chapman 1988; Lisle 1989; Grost et al. 1991).

The amount and size distribution of fine sediment at the surface of channel substrate can affect sediment transport during the incubation period (Carson and Griffith 1987) and thereby affect sedimentation and resultant fine-sediment concentration in redds (Lisle 1989). Within a stream reach, the threshold of stream discharge needed to initiate sediment transport decreases with decreasing particle size at the substrate surface (Carson and Griffiths 1987). Typically, the expected frequency of flow exceedance increases with decreasing stream discharge. Therefore, it is likely that streams with high levels of fine sediment at the substrate surface have a greater frequency and duration of sediment transport of fine sediment than streams with lower levels of surface fine sediment, assuming other factors remain equal. The infiltration of fine sediment into a relatively clean bed substrate in redds appears to be inexorable once sediment transport of fine sediment occurs (Chapman 1988; Lisle 1989). The amount of fine sediment deposited into cleaned gravels is mediated by a variety of factors, but it generally increases as the particle sizes in transport decrease because smaller particles settle more deeply within the substrate (Beschta and Jackson 1979). Thus, it is likely that the amount of fine sediment at the surface of channel substrate may affect salmon survival by influencing the amount of fine sediment in salmon redds during the incubation period.

Despite the biological and management importance of the relationship among ambient substrate conditions, fine sediment levels in redds, and the survival of threatened salmon, there have been no investigations of these relationships in northeastern Oregon, USA and very few elsewhere. The study's objectives were to determine the magnitude and frequency of sedimentation of fine sediment in cleaned gravels in an environment mimicking salmon redds. We also sought to investigate the relationship between surface fine-sediment levels in ambient substrate and the amount of fine sediment that was ultimately deposited within the cleaned gravels in the artificial redds, and the ramifications for the survival of spring chinook within the Grande Ronde River.

Methods

The monitoring of overwinter sedimentation of fine sediment was conducted in the Grande Ronde River during the incubation periods of 1992-1993, 1993-1994, and 1994-1995. Hydrologic and sedimentation results are referenced by the year in which the sample was collected. Overwinter sedimentation was also monitored in spawning habitat during some of these same years in streams in watersheds adjacent to the Grande Ronde River that had lower levels of surface fine sediment to investigate differences in overwinter sedimentation among streams. These streams include Catherine Creek, Clear Creek, and the North Fork of the John Day River (NFJDR). The location of these streams and monitored reaches is shown in Figure 1.

The area of Grande Ronde River watershed above the monitoring locations is about 220 km^2 and ranges in elevation from about 1200 to 2400 m. The watershed is predominantly forested and soils are primarily derived from granitic parent materials. Snow is the dominant form of precipitation and spring snowmelt comprises the bulk of the annual hydrograph. The watersheds of all streams sampled have broadly similar elevations, vegetation, geology, and climate, but the watershed area above the monitored spawning areas, stream size, and upstream land use all vary considerably among watersheds. The watershed areas of Catherine Creek, Clear Creek, and NFJDR are respectively about 260, 90, and 400 km^2. Mean flow width of the streams in monitored reaches during emplacement of samples of Catherine Creek, Clear Creek, NFJDR, and the upper Grand Ronde River were, respectively, about 11.4, 9.15, 21.0, and 8.38 m.

The watershed of the upper Grande Ronde River has been extensively grazed, logged, and roaded over the past 30 years. Portions of the floodplain and river were dredge-mined in the early 1900s (McIntosh et al. 1994). Some sections of the upper Grande Ronde River watershed were burned by wildfire and a subsequent flash flood affected the spawning areas.

The majority of the Catherine Creek watershed is within wilderness. Most of the watershed is grazed. Outside of the wilderness, the watershed has been logged and roaded but to a lesser extent than the Grande Ronde River watershed.

Most of the NFJDR watershed above the sampled reaches is within wilderness. Outside of the wilderness, the watershed has been extensively logged, but to a lesser degree than the upper Grande Ronde River; most of the watershed is grazed by livestock. Some sections of floodplains and the stream have been intensively altered by gravel spoils from historic dredge-mining.

The watershed of Clear Creek has been extensively roaded and logged. Significant portions of the floodplain and stream have been intensively altered by gravel spoils from historic dredge-mining. Most of the watershed is grazed.

Overwinter sedimentation was monitored by placing cleaned gravels in solid-walled containers in spawning habitat at sites constructed to mimic salmon redds. The solid-walled containers were tapered cylinders with an average diameter of 0.102 m and a height of 0.127 m. The redds were constructed in pool tailouts in spawning habitat; the constructed redds had an average area of about 4 m^2 and were designed according to the dimensions described in Bjornn and Reiser (1991). Specialists trained in redd counting provided additional advice on the location and construction of the artificial redds and confirmed that the geometry and size were within the range found in natural salmon redds in the Grande Ronde River (Jeff Zakel, Oregon Dept. of Fish and Wildlife, 107 20th, La Grande, OR 97850, U.S.A.). Three to six artificial redds were constructed in each stream reach monitored. Gravels with diameters > 6.3 mm were taken from the ambient substrate and randomly packed into the containers. Two solid-walled containers of cleaned gravels were emplaced in each constructed redd after the cessation of spawning in the fall and retrieved in the subsequent spring after salmon emergence. The tops of containers were placed about 30 mm below the channel bed surface; a surface layer of gravel covered the containers. The containers were placed in locations within the constructed redd where egg centrums are typically encountered, according to Chapman (1988). However, the egg centrums of spring chinook are typically at depths ranging from 0.2 to 0.3 m (Chapman 1988), while the deepest part of the containers were at a depth of about 0.16 m. The relative merits and precision of this method of sampling fine-sediment accumulation are discussed by Lisle and Eads (1991). Solid-walled containers prohibit lateral infiltration of very fine sediment into cleaned gravels, and, therefore, the amount fine sediment collected in cleaned-gravels solid-walled containers has been considered a minimum estimate of actual amounts (Lisle 1989). Cleaned gravels typically have larger pores than ambient channel substrate; these larger pores tend to increase the depth and amount of infiltration by fine sediment (Lisle 1989). Although Lisle and Eads (1991) suggested that streambed gravels, sieved for fines, and randomly

packed into containers may approximate conditions in redds, it is not known to what extent the gravels placed in the containers deviate from those found in actual redds in the streams we monitored.

We used a particle diameter of < 6.3 mm to define the fine sediment fraction detrimental to salmon survival. However, many descriptors of fine sediment sizes and distribution have been used by a variety of researchers (Young et al. 1991).

The fraction of the streambed covered by fine sediment was ocularly estimated by the technique of Platts et al. (1983) in all monitored reaches during the emplacement and retrieval of samples. Bauer and Burton (1993) noted that ocular estimates of surface fine sediment can have significant observer bias. Therefore, we tested the accuracy and precision of the ocular estimates of the percent of the streambed covered by surface fine sediment against measurements of surface fine sediment by the grid method (Bauer and Burton 1993). The grid method entails placing a sample grid on the channel substrate at equidistant points along a transect across the stream reaches, counting the number of grid intersections that are directly over surface fine sediment and dividing by the total number of intersections to determine the fraction of the surface occupied by fine sediment. In each reach monitored by the grid method, three to five transects were monitored and three to five measurements were taken at each transect. We found that visual estimates of the amount of the substrate surface occupied by fine sediment are relatively accurate and show no consistent bias. The slope of the linear regression line through points of ocularly estimated versus measured surface fine sediment (%) by the grid method was 1.0 and the relationship was statistically significant using a t distribution to test for the significance of the regression slope ($R^2 = 0.92$; $p < 0.01$); the absolute standard error was 5.0%. Due to the accuracy and precision of the ocular estimates, we subsequently dropped measuring surface fine sediment in every monitored reach by the grid method. For the purpose of analysis, individual estimates of surface fine sediment (%) were combined and averaged for each river reach monitored because the mean represents a more areally-integrated description of fine sediment conditions within the reach than the individual estimates at the subreach/transect scale. Fine-sediment accumulations within the solid-walled containers were determined using standard particle size methods. In the Grande Ronde River, streamflow was continuously measured at a gauging station near the sampling points for overwinter sedimentation.

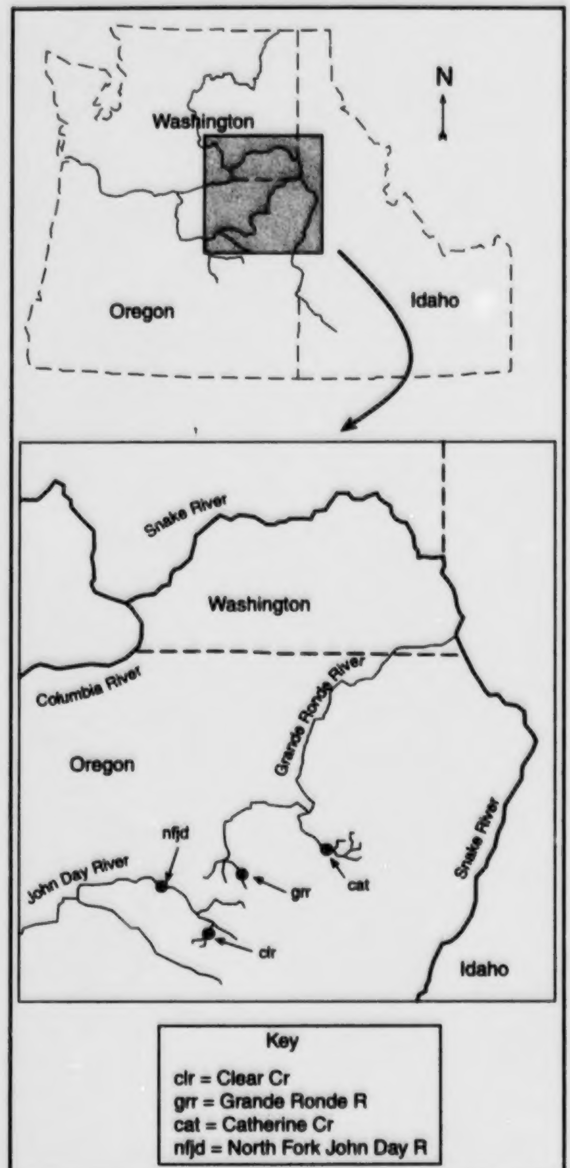

Figure 1. Location map of monitored streams and chinook salmon spawning reaches in northeastern Oregon, U.S.A.

Table 1. Lowest mean, mean mean, and highest mean daily stream discharges measured during the sampling periods in the upper Grande Ronde River, just downstream of sites monitored for overwinter sedimentation

Year	Lowest mean daily flow ($L\ s^{-1}$)	Mean mean daily flow ($L\ s^{-1}$)	Highest mean daily flow ($L\ s^{-1}$)
1993	164	459	2067
1994	312	414	1020
1995	170	534	1473

Table 2. Summary statistics for estimated surface fine sediment and measured overwinter sedimentation in cleaned gravels in solid-walled containers placed in streams

		Estimated surface fines (%)			Fine sediment < 6.3 mm (% by vol.)		
Stream	Year	Mean	SD	n	Mean	SD	n
Gr. Ronde	1993	28.6	3.7	4	18.4	4.1	10
Gr. Ronde	1994	45	8.9	5	18.8	6.8	10
Gr. Ronde	1995	34.5	2.4	5	15.7	4.0	10
Catherine	1994	25	1.8	4	9.9	3.1	8
Catherine	1995	10.4	5.0	5	14.0	4.3	10
NFJDR	1994	20.0	0.0	5	4.9	5.3	9
Clear	1995	13.8	1.2	2	10.5	2.8	4

Results

Streamflow during the sampling periods varied considerably among years in the Grande Ronde River. Table 1 shows the lowest mean daily flow, mean mean daily flow, and highest mean daily flow in the upper Grand Ronde River during the sampling period. Overwinter sedimentation of fine sediment (< 6.3 mm) occurred at variable levels in cleaned gravels in all samples retrieved in all rivers in all years (Table 2; Fig. 2) Overwinter sedimentation of fine sediment in the samples was consistently greatest in the Grande Ronde River which had the highest levels of surface fine sediments (Table 2; Fig. 2). Mean surface fine sediment within monitored reaches did not explain a great deal of the variability in overwinter sedimentation of fine sediment when data from each individual container in all streams and years were analyzed using linear regression, but the relationship is statistically significant (Fig. 2; $R^2 = 0.17$; $p < 0.01$; $n = 30$), using the t distribution to test for significance of the regression slope. When the amount of fine sediment in containers was averaged to yield a mean for each sampled stream for each sampled year, mean surface fine sediment within the sampled reaches explains more of the

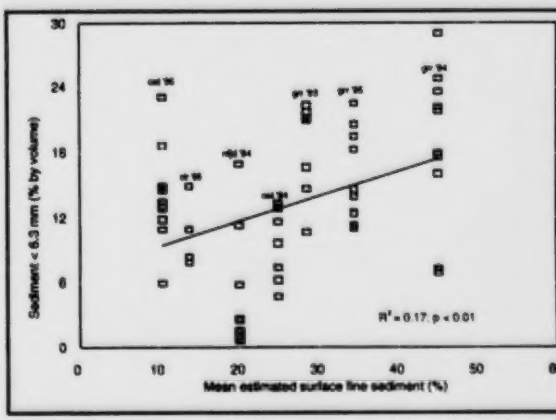

Figure 2. Relationship between all data of measured overwinter sedimentation of fine sediment in individual solid-walled containers and mean estimated surface fine sediment within sampling reaches in all streams collected from 1993 to 1995. cat = Catherine Creek, clr = Clear Creek; grr = upper Grande Ronde River; nfjd = North Fork John Day River.

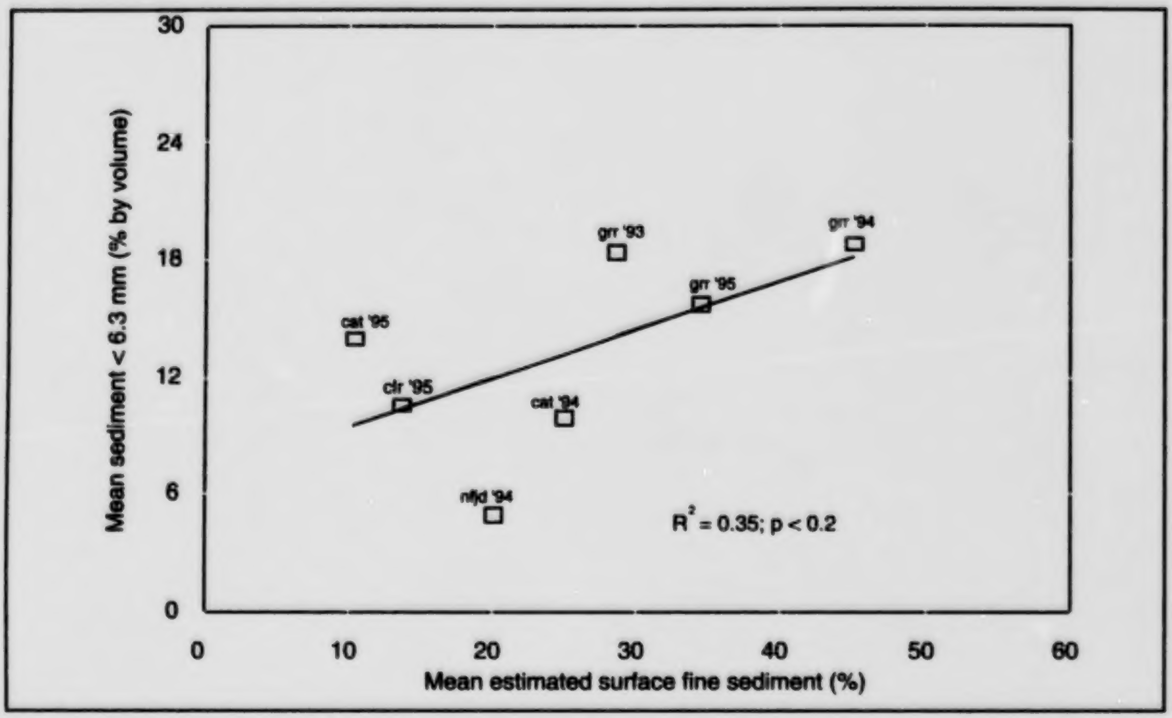

Figure 3. Relationship between mean overwinter sedimentation of fine sediment in containers averaged by stream by year and mean estimated surface fine sediment estimated within sampling reaches. cat = Catherine Creek; clr = Clear Creek; grr = upper Grande Ronde River; nfjd = North Fork John Day River.

variability, but at a lower level of statistical significance (Fig. 3, $R^2 = 0.35$; $p < 0.2$; $n = 7$), than when all individual sample data are analyzed using linear regression and the t distribution to test for significance of the regression slope. Mean surface fine sediment appeared to explain the variability in the overwinter sedimentation of fine sediment among monitored streams in 1994 (Fig. 4; $R^2 = 0.52$; $p < 0.01$), but not in 1995 ($R^2 = 0.08$; $p >> 0.2$), using simple linear regression and using the t distribution to test for significance of the regression slope.

In the upper Grande Ronde River, the magnitude of overwinter sedimentation of fine sediments over the 3 years of monitoring was not related, in a statistically significant fashion, to mean surface fine sediment, mean mean daily flow during the sampling period, or the highest mean daily flow encountered during the sampling period, using linear regression and the *t* distribution to test for significance of the regression slope. We did not use multiple regression with streamflow as an additional independent variable together with surface fine sediment because simple linear regression indicated that streamflow explained almost none of the variability and was not related to overwinter sedimentation in a statistically significant fashion. However, sedimentation of the cleaned gravels in the Grande Ronde may have been at the maximum measurable in the study; visual observation indicated that virtually all voids in the clean gravels were filled by fine sediment. Further, there was little variation in the mean amount of overwinter sedimentation of fine sediment among years in the Grande Ronde (Table 2).

Discussion

Our results clearly indicate that streams in the study area transport fine sediment during the incubation period for spring chinook salmon and that this sediment infiltrates into cleaned gravels in environments similar to salmon redds. The documented increase in fine sediment in cleaned gravels is consistent with other studies in other regions that have shown increases in fine sediment in spawning habitat (Lisle 1989) and salmonid redds (Chapman 1988; Grost et al. 1991) during the incubation period. Our results indicate that although chinook salmon actively winnow fine sediment from redds, this is a transient condition in streams in northeastern Oregon. It

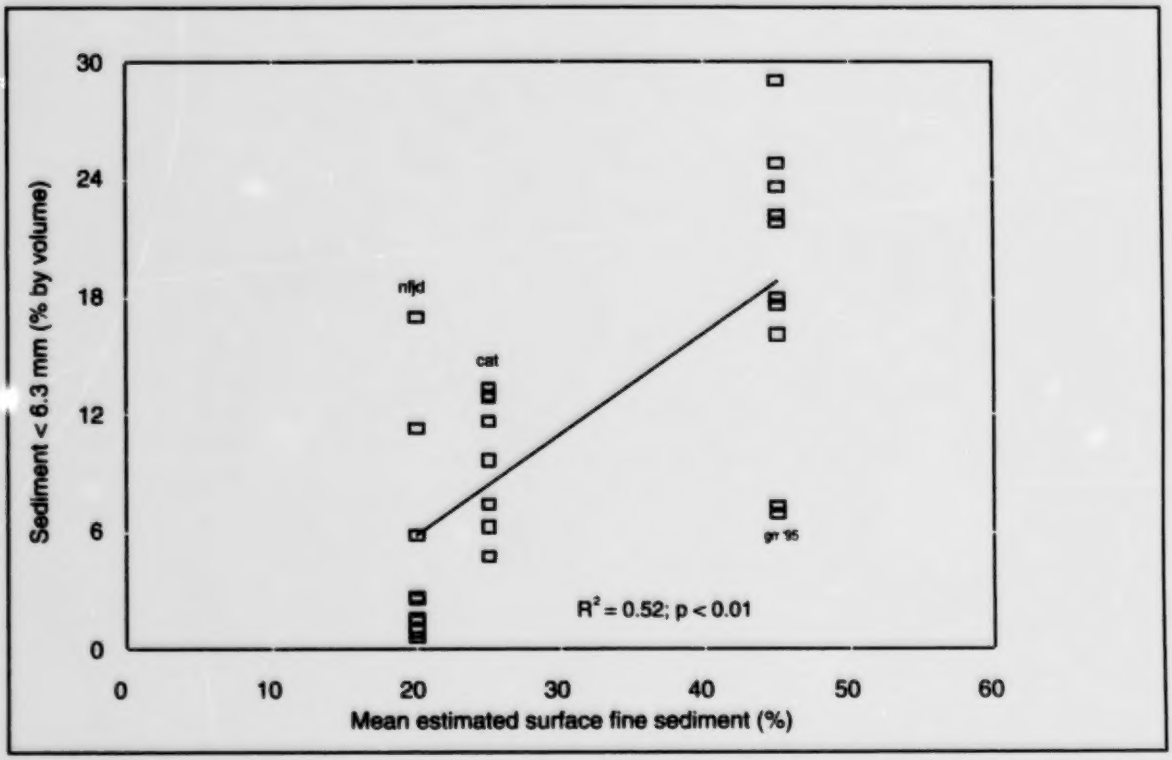

Figure 4. Relationship between measured overwinter sedimentation of fine sediment in containers and mean estimated surface fine sediment in 1994. cat = Catherine Creek; grr = upper Grande Ronde River; nfjd = North Fork John Day River.

cannot be assumed that the active spawning by salmon can completely offset the effects of high levels of fine sediment in ambient substrate during the incubation period.

In the upper Grande Ronde River, the consistently high level of overwinter sedimentation together with the lack of relationship between sedimentation and surface fines may indicate that sediment transport of fine sediment is not limited by the supply of fine sediment that can be entrained at the streamflows that occurred during the sampling periods. This may also explain why overwinter sedimentation amounts in the Grande Ronde do not appear to be related to measures of streamflow magnitude that provide some index of sediment transport capacity during the sampling season, such as mean mean daily flow and highest mean daily flow. Streamflows capable of transporting the relatively abundant sources of fine sediment at the substrate surface appear to occur regularly during the incubation period in the upper Grande Ronde River because the critical discharge needed to transport small diameter sediment is relatively low (Carson and Griffiths 1987). It is likely that salmon survival-to-emergence is reduced the most in streams with high levels of surface sediment, because streams with high amounts of surface fine sediment generally had the highest rates of overwinter sedimentation of fine sediment into cleaned gravels. Studies of the effect of fine sediment on salmonid survival have had somewhat differing results regarding which descriptor of spawning substrate composition best predicts the survival-to-emergence of incubating salmon, the threshold at which survival is significantly reduced, and the precise, functional relationship between substrate composition and salmon survival-to-emergence (Chapman 1988). However, these studies generally have come to the following common conclusions: 1) survival is reduced with increases in fine sediment; 2) relatively small increases in fine sediment can cause relatively large differences in salmonid survival (Chapman 1988; Bjornn and Reiser, 1991; Young et al. 1991). Therefore,

overwinter sedimentation of fine sediment in the Grande Ronde may be a major factor reducing salmon survival-to-emergence. For this reason, it is also valid to attempt to reduce fine sediment levels in the river in order to increase salmon survival and populations.

There are some uncertainties about how our data relate to the actual amount of sedimentation of fine sediment that occurs in egg pockets in natural salmon redds, and resultant survival-to-emergence. The use of gravels in the containers that are completely free of fine sediment may bias the results in opposite directions. Clean gravels tend to have larger pores, which can result in greater rates of intrusion of fine sediment (Beschta and Jackson 1979). On the other hand, in a natural redd environment, intruded fine sediment would be added to the fine sediment initially in the egg pocket, in contrast to our results which only included intruded fine sediment. Research indicates that although salmon winnow fine sediment from the redd, they do not remove all fine sediment (Everest et al. 1987); chinook salmon spawning in western Oregon reduced average fine sediment < 1 mm from 30 to 7.2% in redds after spawning (Everest et al. 1987). Plainly, intruding sediment would be added to the initial fine sediment in the redds, resulting in higher final concentrations of fine sediment than we found in our monitoring, unless the initial fine sediment caused bridging of intruded sediment. Bridging is unlikely because fine sediment remaining in the egg pocket tends to be at depth rather than at the surface of the egg pocket (Chapman 1988). Further, solid-walled containers prevent lateral intrusion which can be a significant component of sedimentation in some substrate environments.

Additionally, the placement of the containers within the artificially constructed redds may also bias the results. The bottoms of the containers were generally at a depth of about 0.16 m. In streams in Idaho with similar hydrology and stream substrate composition to our study area, the greatest increase in fine sediment within artificial redds occurred at a depth of about 0.30 m (Russ Thurow, unpublished data, U.S. Forest Service, Intermountain Research Station, 316 Myrtle St., Boise, ID, 83702). For these reasons, the amount of fine sediment that we found in our containers may be the minimum that intrudes into natural egg pockets in salmon redds in the study area.

There is also some uncertainty regarding the severity of the reductions in salmon survival caused by the levels of overwinter sedimentation that we measured. Previous research has indicated relatively small decreases in survival with increasing levels of fine sediment < 6.3 mm in diameter up to about 20% fine sediment by volume (Bjornn and Reiser 1991). However, in the samples from 1993 and 1994, > 90% of the fine sediment that had intruded into the containers was < 2 mm in diameter. Reiser and White (1988) found that egg survival dropped significantly as the concentration of fine sediment with diameters ranging from 0.84 to 4.6 mm increased; at 20% fine sediment with diameters ranging from 0.84 to 4.6 mm, chinook salmon survival was about 60% of the survival level when fine sediment concentration was zero. This relationship was even more precipitous as the concentration of fine sediment < 0.84 mm increased. Therefore, if our data accurately represent the final concentrations of fine sediment in redds due to overwinter sedimentation, it is likely that fine sediment is causing significant decreases in salmon survival. Actual decreases in salmon survival may be even more severe because the sampling methods tend to bias results towards lower final concentrations of fine sediment in the containers.

In the range of surface fine sediment conditions encountered every level confers some risk of reduced salmon survival-to-emergence by overwinter sedimentation in some sites. For instance, relatively high levels of overwinter sedimentation occurred in some of the samples in 1995 in Catherine Creek even though it had the lowest mean estimated surface fine sediment levels in our study (Table 2; Fig. 1).

The relatively high level of variability in the amount of sedimentation by fine sediment within and among streams and from year to year is not surprising given the number of factors that probably influence the intrusion of fines into clean gravels. Overwinter sedimentation within redds is influenced by both the characteristics of sediment transport over the redd and factors that control sediment deposition in substrate. Factors influencing the transport of streambed sediment operate at a variety of scales and include stream discharge, channel morphology attributes that influence flow depth, velocity, and turbulence, particle sizes at the bed surface, and other factors, such as bed roughness and particle size distribution (Carson and Griffiths 1987). Sedimentation within cleaned gravels is probably influenced by the magnitude and duration of sediment transport over the cleaned gravels, particle sizes in transport over the gravels, and the size and volume of the pores in the cleaned gravels (Lisle

1989). Most of these physical factors and processes influencing sedimentation have considerable spatial and temporal variability. Thus, overwinter sedimentation in actual redds is expected to be highly variable, consistent with our data.

Considering the complexity and variability of the factors influencing the sedimentation of fine sediment in cleaned gravels, it is not surprising that the surface fine sediment (%) alone did not explain a great deal of the variability when all data on overwinter sedimentation were analyzed. While the basic characteristics of sediment transport remain difficult to reliably predict, enough is known to safely state that no single, easily measurable channel attribute will explain most of the variability in sedimentation of fine sediment in cleaned gravels or salmon redds. Detailed monitoring of the physical factors and processes influencing sediment transport might have provided data to explain more of the variability in sedimentation of fine sediment. However, such efforts are relatively expensive, onerous and are not feasible as part of a program of routine monitoring of conditions in salmon habitat. In contrast, estimation of surface fine sediment can be done quickly, is inexpensive, and appears to be related to overwinter sedimentation in cleaned gravels in the field environment in a statistically significant fashion. It appears that surface fine sediment levels in ambient substrate provide an index of the susceptibility of redds to significant overwinter sedimentation and reduced salmon survival within spawning habitat in streams.

Acknowledgments

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Streams and Salmonid Assemblages within Roaded and Unroaded Landscapes in the Clearwater River Sub-basin, Idaho

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Abstract

From 1989 through 1995, stream conditions and salmonid abundance were surveyed along 1769 km of streams on national forest lands within the Clearwater River sub-basin, Idaho. Fish habitat and riparian conditions were evaluated using a transect-based method, while fish populations were sampled at representative locations (998 total) along the streams examined. Analyses of the data collected suggest that there are important differences between streams within roaded and unroaded landscapes. Streams in unroaded areas had higher quality habitat with significantly lower levels of fine sediment than those in roaded landscapes despite a relatively recent (<100 yr) history of catastrophic wildfires in many unroaded watersheds. The higher quality habitat in unroaded areas tended to support more diverse and abundant populations of native trout, particularly in lower gradient channels where the most pronounced cumulative effects of past disturbance might be expected. This paper examines these and other differences between streams in roaded and unroaded landscapes, briefly discussing their significance in the context of ecosystem management and ongoing efforts to conserve sensitive aquatic species.

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Introduction

Aquatic ecosystems in the Columbia River Basin (CRB) have been affected by a variety of natural and anthropogenic disturbances since Euro-American settlement. Catastrophic wildfires, road construction, logging, mining, livestock grazing, water development, introductions of exotic species, or other human activities have had deleterious effects on many of the region's aquatic systems (McIntosh et al. 1994; Rhodes et al. 1994; Wissmar et al. 1994) . Sensitive aquatic species dependent upon these systems are in decline across much of the CRB (Nehlsen et al. 1991; Henjum et al. 1994; Lee et al. 1997) and several have received or been considered for federal protection under the Endangered Species Act of 1973.

Land managers in the CRB are now developing strategies for resource conservation that focus greater attention on disturbance regimes, recovery processes, and maintaining natural variation in ecosystem conditions. These strategies will require detailed information on natural patterns of variation in streams, and on broad-scale differences in how streams within human-altered landscapes may differ from those in relatively pristine areas. This paper provides information in response to these issues for streams draining a large national forest in the Clearwater River sub-basin, Idaho, a diverse landscape that has some of the most extensive unroaded areas remaining within the CRB.

Methods

Stream Inventory

From 1989 through 1995, fish habitat and populations were surveyed along 2074 distinct reaches of 313 streams scattered across the 7435 km^2 Clearwater National Forest (CNF), the dominant landholder within the northern half of the Clearwater River sub-basin, Idaho (Fig. 1). The surveys, part of an extensive inventory program funded by CNF, covered a total of 1769 km of fish-bearing or potentially fish-bearing streams. Streams in the CNF typically have watersheds dominated by highly erodible batholithic granites, are affected by a diversity of natural and anthropogenic disturbances, and flow from designated wilderness, unroaded areas, or roaded watersheds that are subjected to varying levels of disturbance. Over 60% of the CNF is roadless, with 15% of its land base afforded the

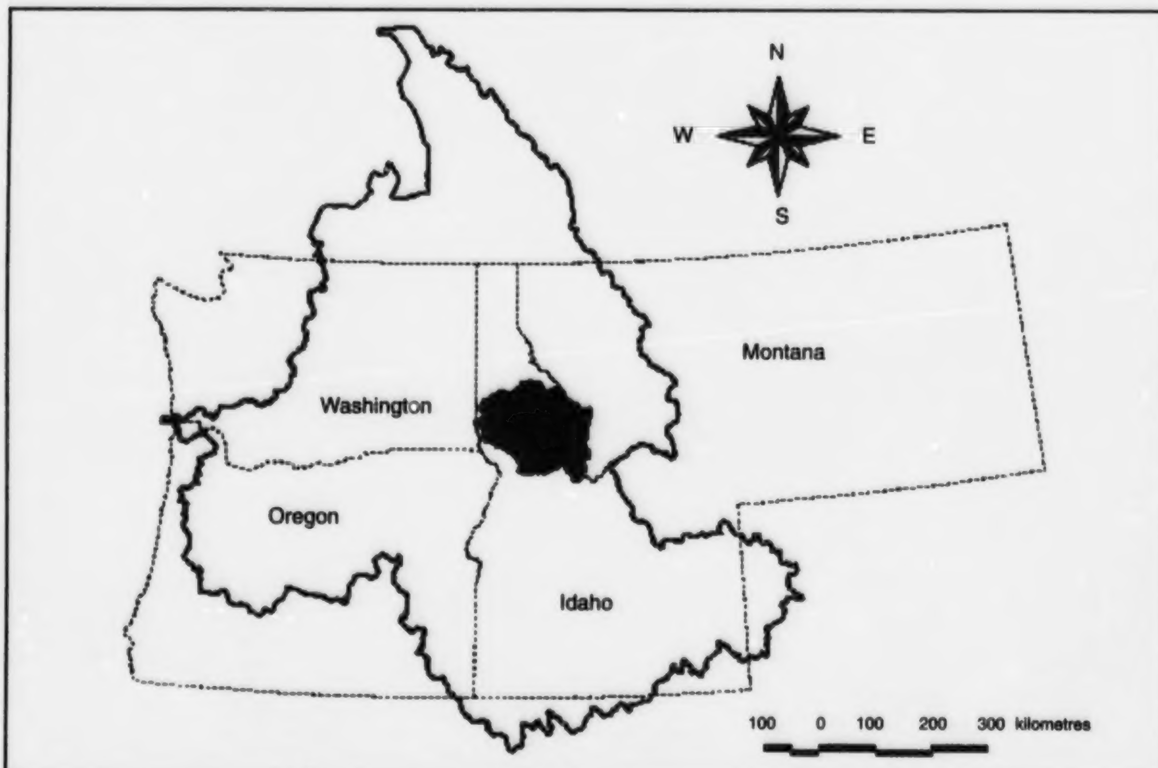

Figure 1. Location map for the Clearwater River sub-basin, Idaho, within the larger Columbia River Basin.

protection of federal wilderness designation and another 8% recommended for such designation (Clearwater National Forest 1988).

Each stream surveyed was broken into distinctive reaches based on channel types (Rosgen 1985) and the locations of major tributary junctions. Habitat conditions within these reaches were then evaluated using Espinosa's (1988) modification of transect-based methods described by Platts et al. (1983). Specific habitat features, including coarse woody debris, pools, and cobble embeddedness, were examined at constant 30 m or, occasionally, 60 m intervals along each reach of stream surveyed. Two forms of coarse woody debris were measured. Acting debris, defined as stable woody material at least 10 cm in diameter that had an independent and direct instream effect upon fish habitat, was counted and expressed as number of pieces per 100 m of stream. Potential debris, the number of standing trees or snags along the stream which were at least 30.5 cm diameter at breast height and capable of falling into the stream to become acting debris, was measured with a slope-compensating angle gauge and expressed as the number of pieces per 100 m. Cobble embeddedness was estimated at each habitat transect, and later adjusted using regression equations developed from paired-estimate and measurement data from systematically selected calibration transects. Cobble embeddedness at the calibration transects was measured as described by Torquemada and Platts (1988).

Fish populations were sampled at 998 representative 30 to 60 m stations distributed among the stream reaches surveyed. Estimates of the species composition and abundance of salmonids at most of these stations were direct snorkel counts (Griffith 1981; Platts et al. 1983), the standard method in batholith streams with high water clarity and low conductivity (Petrosky and Holubetz 1987). Estimates were made using electrofishing gear and standard removal-depletion methods (Zippin 1958) at 113 stations where water clarity was too low for effective snorkel counts.

Analytical Techniques

Analyses of data collected along the surveyed reaches and streams were summarized in multiple reports to the CNF (Clearwater BioStudies, Inc. 1989, 1991, 1992, 1993, 1994, 1995, 1996). Mean habitat conditions reported for each stream reach, and abundances (no./100 m^2) of salmonids found at each station sampled, were subsequently compiled into databases stratified by landscape treatment (roaded versus unroaded) and four major channel types (Aa, A, B, or C; Table 1).

I then compared the abundance of coarse woody debris and pools, and levels of cobble embeddedness, in roaded versus unroaded reaches of each of the four major channel types surveyed. The comparisons were based both on measures of central tendency (mean or median) and on landscape-level patterns of variation of these habitat features within each treatment and channel type. Statistical tests of differences in median conditions between roaded and unroaded reaches of each channel type were made using non-parametric Mann-Whitney rank sum tests. Patterns of variation were assessed by examining ranges of parameter values and by inspecting exceedance curves that depicted the cumulative frequencies of habitat conditions found in roaded versus unroaded reaches of each channel type. The exceedance curves were similar to those the US Geologic Survey (Riggs 1968) utilizes when assessing variability in streamflows.

Trout populations at stations within roaded versus unroaded landscapes were also compared. These comparisons examined similarities and differences between landscape treatments in terms of trout assemblages, abundances of overyearling trout and patterns of species occurrence within each major channel type. Statistical tests of differences in median trout abundances between stations in roaded and unroaded reaches of each channel type were made using non-parametric Mann-Whitney rank sum tests. Landscape-level patterns of variation in trout abundance, and differences in those patterns between landscape treatments, were evaluated using exceedance curves similar to those I developed to assess variability in habitat conditions.

Results

Before describing current differences between streams in the CNF's roaded and unroaded portions of the Clearwater River sub-basin, I should note that differences likely existed between them even before roads were constructed in the managed areas. This is due in part to a pattern of development which has not been evenly distributed across all geomorphic features or histories of natural disturbance. With regard to historic disturbances, a high proportion of the CNF's unroaded watersheds are still slowly recovering from catastrophic forest fires that occurred during the early 1900s. For this reason, many or most of the streams surveyed in unroaded areas were still recovering from historic disturbance and exhibited aquatic habitat conditions that were

Table 1. Strata for stream reaches and fish stations sampled in the Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95

Strata	Descriptions
Landscape treatments	
Roaded	streams draining watersheds with roaded character, ranging from systems with very modest road networks (e.g., road densities more than about 0.05 km/km^2) to drainages with very high road densities
Unroaded	streams draining watersheds that either lacked roads or had such limited road networks that their character was essentially unroaded
Channel types (Rosgen 1985)	
Aa	extremely steep (>10.0% gradient), strongly confined streams with low sinuosity
A	steep (4.0–10.0% gradient), strongly confined streams with low sinuosity
B	moderately steep (1.5–4.0%), moderately confined, and moderately sinuous streams
C	low gradient (<1.5%), slightly confined, and highly sinuous streams

Note: E- and G-type streams (Rosgen 1994) surveyed after 1992 were grouped with C- and B-type streams, respectively, because that is how those two types were classified in the earlier years.

below the streams' potentials. Fish habitat in the unroaded streams, therefore, did not necessarily represent an optimal condition against which to measure the degree to which other streams may have been affected by natural or anthropogenic disturbances. Because much of the currently roaded landscape on the CNF was unaffected by the catastrophic fires which burned many watersheds in the early 1900s, streams in roaded areas may, on average, have been in better condition (relative to potential) before their watersheds were first roaded than many streams in unroaded areas are today.

Another key distinction between the CNF's roaded and unroaded landscapes is that cumulative land-use effects have not been evenly distributed across the stream channel types present. This is important because certain aquatic animals may prefer specific types of channels. For example, spring chinook salmon (*Oncorhynchus tshawytscha*) tend to prefer slow-flowing habitat prevalent in C-type channels which are sensitive to disturbance. Of the surveyed reaches, 39% of Aa channels, 42% of A channels, 53% of B channels and 60% of C channels have been classified as roaded (Fig. 2). Nearly all of the unconstrained, low-gradient C channels not classified as roaded were found in large watersheds that lacked roads. Relatively few C channels were found in small unroaded watersheds.

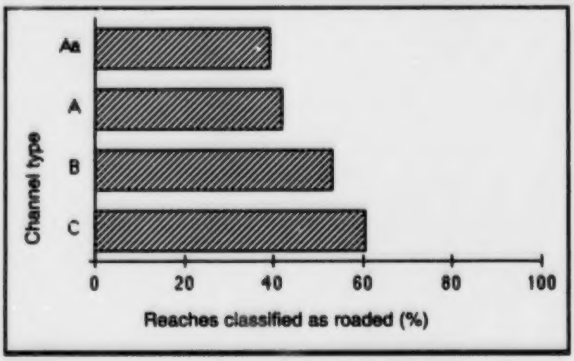

Figure 2. Percent of surveyed stream reaches classified as roaded, Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95.

Fish Habitat

Coarse Woody Debris.

Levels of coarse woody debris along the reaches surveyed were affected by interactions between stream potentials, uneven anthropogenic impacts, and a mosaic of natural, ongoing fire recovery processes. In both managed and unroaded landscapes, acting and potential woody debris were most variable and abundant in steep, constrained Aa and A-type channels and less abundant along lower-gradient, less-constrained channels (Table 2; Figs. 3

Table 2. Values of selected habitat parameters, by channel type and landscape treatment, for stream reaches in the Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95

Habitat parameter	Channel type	Landscape[a]	Range of variation	Mean (SE)	Median	P[b]
Acting debris (pieces/100 m)	Aa	RD	1 - 120	23.9 (1.20)	21.2	
		UN	0 - 75	19.9 (0.87)	17.4	<0.001
	A	RD	0 - 75	18.3 (0.72)	17.8	
		UN	0 - 78	14.2 (0.62)	11.9	<0.001
	B	RD	0 - 41	10.4 (0.46)	8.0	
		UN	0 - 55	11.1 (0.65)	7.9	0.418
	C	RD	0 - 55	10.9 (0.71)	7.0	
		UN	0 - 54	16.9 (1.04)	15.5	<0.001
Potential debris (pieces/100 m)	Aa	RD	0 - 136	42.2 (1.99)	41.0	
		UN	0 - 203	31.3 (1.44)	26.2	<0.001
	A	RD	0 - 157	33.0 (1.47)	28.0	
		UN	0 - 171	28.2 (1.09)	22.0	0.013
	B	RD	0 - 104	26.8 (1.16)	23.0	
		UN	0 - 131	24.2 (1.10)	19.4	0.212
	C	RD	0 - 86	15.7 (1.03)	11.3	
		UN	0 - 85	21.6 (1.25)	18.0	<0.001
Percent pool	Aa	RD	0 - 55	19.7 (0.90)	18.0	
		UN	0 - 100	26.4 (1.12)	23.0	<0.001
	A	RD	0 - 100	21.9 (0.86)	19.7	
		UN	0 - 100	24.3 (0.85)	21.0	0.170
	B	RD	0 - 71	18.7 (0.77)	16.0	
		UN	0 - 98	22.5 (1.01)	18.0	0.007
	C	RD	0 - 100	41.0 (1.96)	34.2	
		UN	0 - 100	44.8 (2.16)	44.0	0.107
Percent cobble embeddedness	Aa	RD	0 - 100	41.4 (1.37)	38.0	
		UN	0 - 85	30.3 (1.14)	27.2	<0.001
	A	RD	4 - 100	40.4 (1.23)	37.0	
		UN	3 - 100	28.2 (0.93)	24.0	<0.001
	B	RD	4 - 100	42.9 (1.43)	34.0	
		UN	3 - 96	28.3 (1.02)	24.0	<0.001
	C	RD	6 - 100	71.6 (2.00)	83.0	
		UN	4 - 100	53.0 (2.22)	52.8	<0.001

[a] Sample sizes were 172 Aa-, 277 A-, 328 B-, and 213 C-type reaches in roaded (RD) landscapes, and 266 Aa-, 388 A-, 288 B-, and 140 C-type reaches in unroaded (UN) landscapes.

[b] Significance of differences in median abundances between landscapes, based on non-parametric, Mann-Whitney rank sum tests.

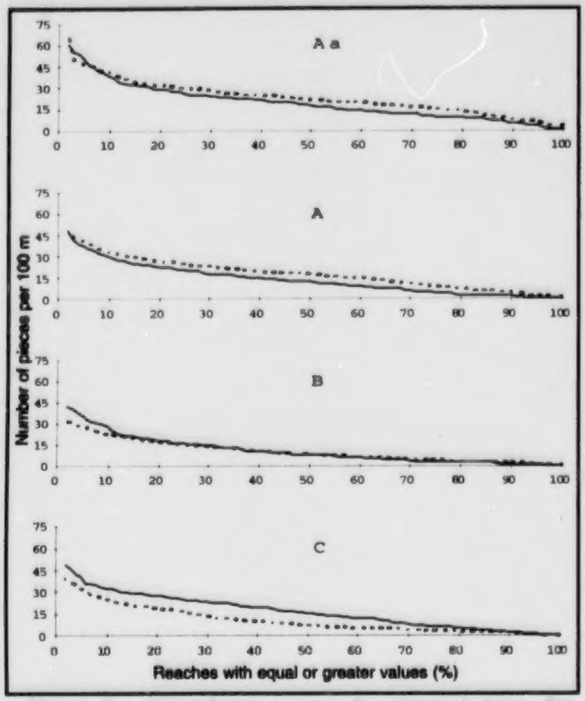

Figure 3. Exceedance curves, by channel type, for acting woody debris in stream reaches within roaded (dashed lines) and unroaded (solid lines) landscapes, Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95. Sample sizes were 172 Aa-, 277 A-, 328 B-, and 213 C-type reaches in roaded landscapes, and 266 Aa-, 388 A-, 288 B-, and 140 C-type reaches in unroaded landscapes.

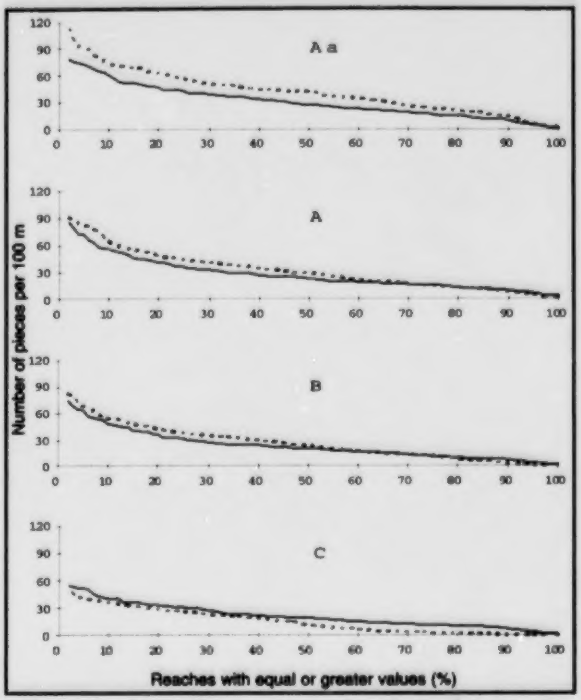

Figure 4. Exceedance curves, by channel type, for potential woody debris along stream reaches within roaded (dashed lines) and unroaded (solid lines) landscapes, Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95. Sample sizes were 172 Aa-, 277 A-, 328 B-, and 213 C-type reaches in roaded landscapes, and 266 Aa-, 388 A-, 288 B-, and 140 C-type reaches in unroaded landscapes.

and 4). Despite these similarities, however, median levels of both acting and potential debris were significantly lower in unroaded than in roaded Aa ($p<0.001$) and A channels ($p<0.01$). Between-treatment differences in woody debris abundance were relatively small among B channels, but unroaded C channels had significantly more ($p<0.001$) acting and potential debris than did managed ones.

Pools.

Pool abundance (expressed as percent pool) ranged from extremely low to very high among the reaches surveyed, with the greatest abundance in both landscape types typically found in low-gradient, C-type channels (Table 2; Fig. 5). Ranges of variation in pool abundance were either greater in unroaded landscapes or similar among landscapes, depending on channel type. Median abundances of pool habitat were higher in unroaded than managed stream reaches of each major channel type, although these differences between landscapes were statistically significant only for Aa ($p<0.001$) and B-type channels ($p = 0.007$).

Cobble Embeddedness.

There were only minor differences in the ranges of variation in cobble embeddedness exhibited among stream reaches within managed and unroaded landscapes. For each channel type and landscape treatment, at least a small proportion of the surveyed reaches exhibited either relatively low or extremely high embeddedness (Table 2; Fig. 6). However, low levels of embeddedness were less common and high levels far more common in managed than in unroaded

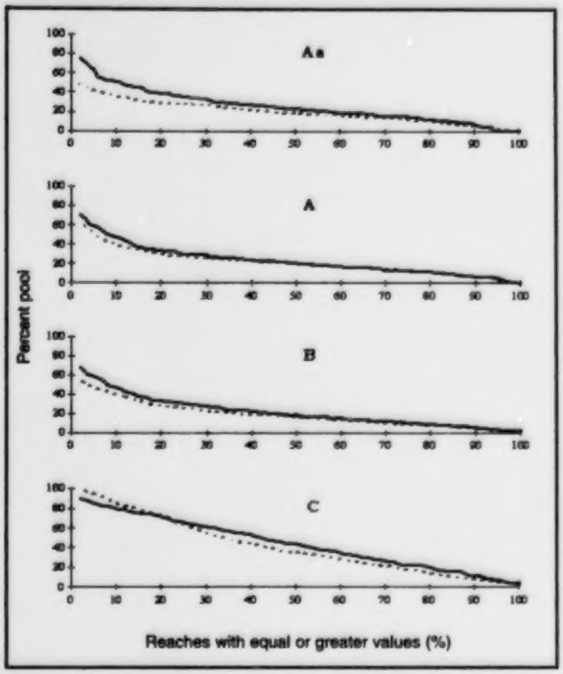

Figure 5. Exceedance curves, by channel type, for percent pool habitat in stream reaches within roaded (dashed lines) and unroaded (solid lines) landscapes, Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95. Sample sizes were 172 Aa-, 277 A-, 328 B-, and 213 C-type reaches in roaded landscapes, and 266 Aa-, 388 A-, 288 B-, and 140 C-type reaches in unroaded landscapes.

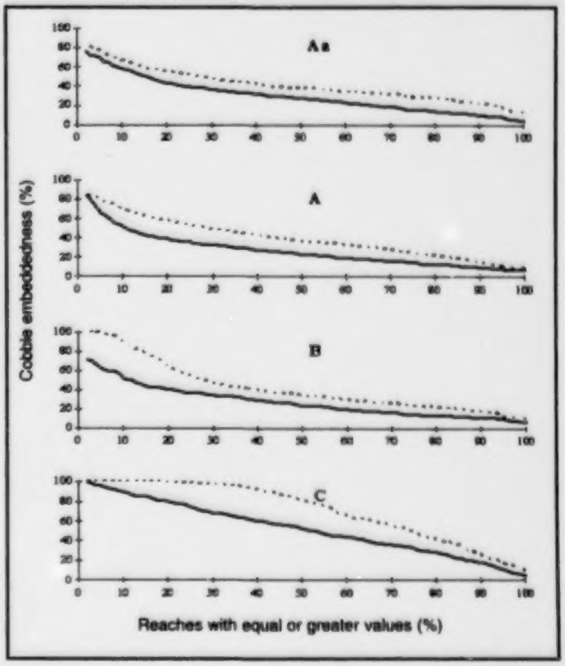

Figure 6. Exceedance curves, by channel type, for the cobble embeddedness of stream reaches within roaded (dashed lines) and unroaded (solid lines) landscapes, Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95. Sample sizes were 172 Aa-, 277 A-, 328 B-, and 213 C-type reaches in roaded landscapes, and 266 Aa-, 388 A-, 288 B-, and 140 C-type reaches in unroaded landscapes.

areas. In spite of high sediment levels that have persisted in some unroaded streams as after-effects of historic fires, cobble embeddedness was significantly higher ($p<0.001$) within managed landscapes than in unroaded areas. This pattern held for each channel type, and was most pronounced in the low-gradient C channels, where median embeddedness in managed reaches (83.0%) was far higher than in unroaded ones (52.8%).

Trout Assemblages

Salmonids in the surveyed streams included rainbow-steelhead trout (*O. mykiss*), westslope cutthroat trout (*O. clarki lewisi*), bull trout (*Salvelinus confluentus*), introduced brook trout (*S. fontinalis*), spring chinook salmon and mountain whitefish (*Prosopium williamsoni*). Each of these species was present within both managed and unroaded landscapes. My analysis focused on trout found in the streams, because chinook salmon were relatively abundant only in a few C- and B-type channels where their presence was often strongly influenced by hatchery supplementation. Whitefish were observed only at a very small proportion of the stations sampled.

Native Trout.

Collectively, native species of trout occupied higher percentages of the stations sampled in A and B channels than in Aa or C channels (Fig. 7). This pattern held in both managed and unroaded landscapes, as did a predominance of cutthroat-only and cutthroat-rainbow trout assemblages. Despite these similarities, however, native trout occupied smaller percentages of the stations within roaded landscapes than they did in unroaded areas. Differences in the composition of native trout assemblages were greatest between roaded and unroaded C channels, where

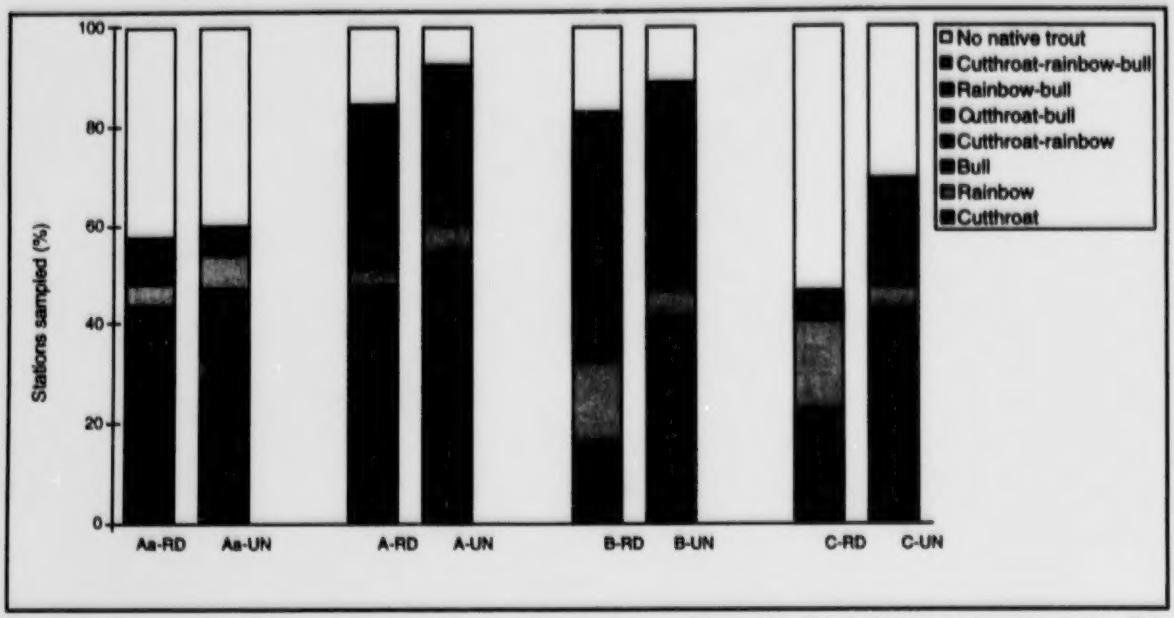

Figure 7. Frequencies of occurrence for various native trout assemblages, by channel type, at representative fish stations sampled in roaded (RD) and unroaded (UN) landscapes, Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95. Sample sizes were 71 Aa-,174 A-, 158 B-, and 136 C-type stations in roaded landscapes, and 92 Aa-, 185 A-, 129 B-, and 53 C-type stations in unroaded landscapes.

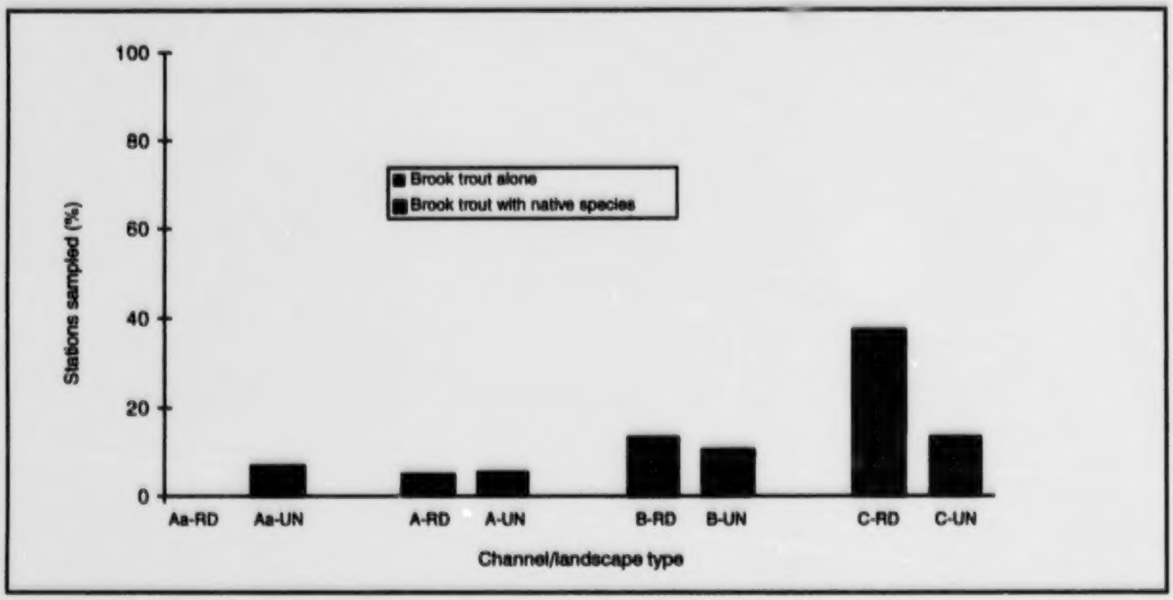

Figure 8. Frequencies of occurrence for trout assemblages that included brook trout, by channel type, at representative fish stations sampled in roaded (RD) and unroaded (UN) landscapes, Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95. Sample sizes were 71 Aa-, 174 A-, 158 B-, and 136 C-type stations in roaded landscapes, and 92 Aa-, 185 A-, 129 B-, and 53 C-type stations in unroaded landscapes.

one-species (40% v. 47%), two-species (6% v. 15%), and three-species complexes (1% v. 8%) were each observed at a lower percentage of stations in roaded areas.

Brook Trout.

Introduced brook trout were found at stations in both roaded and unroaded landscapes, but tended to be more widely distributed in roaded areas, where there were more abundant opportunities for their introduction (Fig. 8). In roaded landscapes, brook trout were found in multiple directions from known release points that were frequently accessible by road. Brook trout observed in streams within unroaded landscapes were found almost exclusively in areas downstream from historic outplants into headwater lakes. Patterns of this species' occurrence in both managed and unroaded areas strongly suggest that it has been more successful invading low to moderate gradient channels (C or B types) than steep, constrained ones (A or Aa types). Brook trout also seem to have been most successful in becoming the dominant salmonid where CNF streams have highly embedded substrates (C. Huntington, unpublished data).

Patterns of Abundance of Overyearling Trout

The abundance of age 0 trout is highly variable in streams and can often mask major differences in the abundance of older-aged fish. For this reason, abundances of overyearling fish (i.e., those at least one-year old) in streams were assumed to be better indicators of population responses to habitat conditions than the combined abundances of trout of all ages.

Native Trout.

Overyearling trout of all native species combined were generally more abundant, more widely distributed among stations, and exhibited greater variation in abundance, within unroaded areas than in managed landscapes (Table 3; Fig. 9). These patterns held for each channel type, and were pronounced in B and C channels, where mean abundances at stations were 35% (6.5 v. 4.4 fish/100 m^2) and 232% (6.3 v. 1.9 fish/100 m^2) higher, respectively, in unroaded areas.

Cutthroat trout were more abundant and widely distributed among the stations sampled than were rainbow-steelhead or bull trout (Figs. 10-12). This was true for all four major channel types and within both managed and unroaded landscapes.

Overyearling cutthroat trout were generally present at a higher percentage of stations, and at higher levels of abundance, in unroaded than in managed B- and C-type channels. These differences were greatest in C-type channels, where overyearling cutthroat were present at more than twice as high a percentage of stations (63% v. 29%) and at mean abundances more than three times as high (5.7 v. 1.4 fish/100 m^2) in unroaded areas. However, there was little overall difference in the frequency of occurrence or abundance of overyearling cutthroat trout between stations in unroaded and managed A-type channels, where they tended to be most abundant.

The distribution and abundance of rainbow-steelhead trout within the CNF during this study was affected by poor escapements of spawners due to high mortality rates downstream at hydroelectric projects and in fisheries. Differences between the median abundances of these fish at stations in roaded versus unroaded landscapes were statistically insignificant in all four channel types. However, overyearling rainbow-steelhead trout occurred most frequently and were most abundant at stations in A- and B-type channels (Table 3), within which they tended to reach their highest levels of abundance in unroaded areas (Fig. 11).

Bull trout were the least common of the three species of native trout found at the stations sampled, and observed differences in their abundance between managed and unroaded landscapes were relatively small (Table 3; Fig. 12). Juvenile bull trout were observed only in a few CNF watersheds, giving the clear impression that the species' spawning distribution is strongly skewed toward very small portions of the managed and unroaded landscapes. Known spawning areas for bull trout within the CNF are only in the coldest streams and reaches examined (C. Huntington, unpublished data).

Brook Trout.

Unlike the three species of trout native to the study area, brook trout were most abundant and occurred most frequently at stations sampled in low-gradient, C-type channels (Table 4; Fig. 13). These were the only channels where brook trout might be considered relatively common in the CNF. The mean abundance of overyearling brook trout in C-type channels was second only to overyearling cutthroat trout in unroaded areas and matched that of cutthroat trout within roaded landscapes. Regardless of channel type or landscape treatment, brook trout tended to be the most abundant species of trout at stations where they occurred (C. Huntington, unpublished data).

Table 3. Abundance (no./100 m²) of over-yearling native trout, by channel type and landscape treatment, at stations in the Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95

Species	Channel type	Land-scape[a]	Range of variation	Mean (SE)	Median	P[b]
All native species	Aa	RD	0.0 - 25.8	4.5 (0.68)	1.6	
		UN	0.0 - 32.1	5.9 (0.75)	3.2	0.347
	A	RD	0.0 - 44.4	9.0 (0.64)	6.9	
		UN	0.0 - 71.2	9.6 (0.68)	7.6	0.424
	B	RD	0.0 - 29.4	4.8 (0.49)	2.9	
		UN	0.0 - 31.0	6.5 (0.52)	4.9	<0.001
	C	RD	0.0 - 30.1	1.9 (0.38)	0.0	
		UN	0.0 - 37.9	6.3 (1.08)	4.2	<0.001
Cutthroat trout	Aa	RD	0.0 - 25.8	3.8 (0.66)	1.3	
		UN	0.0 - 25.4	5.0 (0.71)	1.3	0.513
	A	RD	0.0 - 41.0	7.5 (0.62)	5.4	
		UN	0.0 - 71.2	7.0 (0.67)	4.6	0.499
	B	RD	0.0 - 26.8	3.0 (0.22)	0.9	
		UN	0.0 - 31.1	4.7 (0.49)	3.2	<0.001
	C	RD	0.0 - 30.1	1.4 (0.37)	0.0	
		UN	0.0 - 37.9	5.7 (1.01)	1.9	<0.001
Rainbow trout	Aa	RD	0.0 - 13.1	0.6 (0.26)	0.0	
		UN	0.0 - 28.5	0.9 (0.27)	0.0	0.856
	A	RD	0.0 - 10.4	1.3 (0.22)	0.0	
		UN	0.0 - 13.6	2.4 (0.37)	0.0	0.191
	B	RD	0.0 - 19.4	1.7 (0.22)	0.0	
		UN	0.0 - 17.3	1.8 (0.31)	0.0	0.058
	C	RD	0.0 - 9.0	0.6 (0.13)	0.0	
		UN	0.0 - 17.8	0.5 (0.18)	0.0	0.994
Bull trout	Aa	RD	0.0 - 2.3	0.0 (0.03)	0.0	
		UN	0.0 - 0.6	0.0 (0.01)	0.0	0.972
	A	RD	0.0 - 7.2	0.2 (0.06)	0.0	
		UN	0.0 - 6.4	0.1 (0.05)	0.0	0.637
	B	RD	0.0 - 6.9	0.1 (0.05)	0.0	
		UN	0.0 - 1.8	0.1 (0.03)	0.0	0.728
	C	RD	0.0 - 0.8	0.0 (0.01)	0.0	
		UN	0.0 - 1.1	0.1 (0.03)	0.0	0.261

[a] Sample sizes were 71 Aa-, 174 A-, 158 B-, and 136 C-type stations in roaded (RD) landscapes, and 92 Aa-, 185 A-, 129 B-, and 53 C-type stations in unroaded (UN) landscapes.

[b] Significance of differences in median abundances between landscapes, based on non-parametric, Mann-Whitney rank sum tests.

Table 4. Abundance (no./100 m²) of over-yearling brook trout, by channel type and landscape treatment, at stations in the Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95

Species	Channel type	Land-scape[a]	Range of variation	Mean (SE)	Median	P[b]
Brook trout	Aa	RD	0.0 - 0.0	0.0 (0.00)	0.0	
		UN	0.0 - 32.0	0.7 (0.39)	0.0	0.476
	A	RD	0.0 - 34.4	0.5 (0.25)	0.0	
		UN	0.0 - 5.0	0.1 (0.04)	0.0	0.987
	B	RD	0.0 - 18.8	0.5 (0.16)	0.0	
		UN	0.0 - 23.7	0.5 (0.25)	0.0	0.962
	C	RD	0.0 - 15.6	1.4 (0.23)	0.0	
		UN	0.0 - 40.5	2.5 (1.10)	0.0	0.037

[a] Sample sizes were 71 Aa-, 174 A-, 158 B-, and 136 C-type stations in roaded (RD) landscapes, and 92 Aa-, 185 A-, 129 B-, and 53 C-type stations in unroaded (UN) landscapes.

[b] Significance of differences in median abundances between landscapes, based on non-parametric, Mann-Whitney rank sum tests.

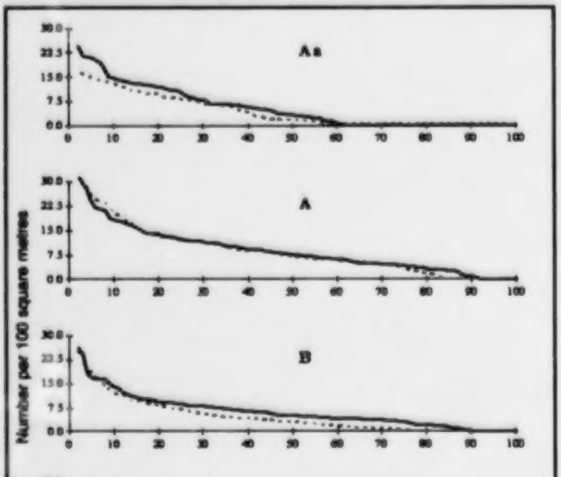

Figure 9. Exceedance curves, by channel type, for the abundance (no./100 m²) of overyearling native trout at stations in roaded (dashed lines) and unroaded (solid lines) landscapes, Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95. Sample sizes were 71 A-a, 174 A-, 158 B-, and 136 C-type stations in roaded landscapes, and 92 Aa-, 185 A-, 129 B-, and 53 C-type stations in unroaded landscapes.

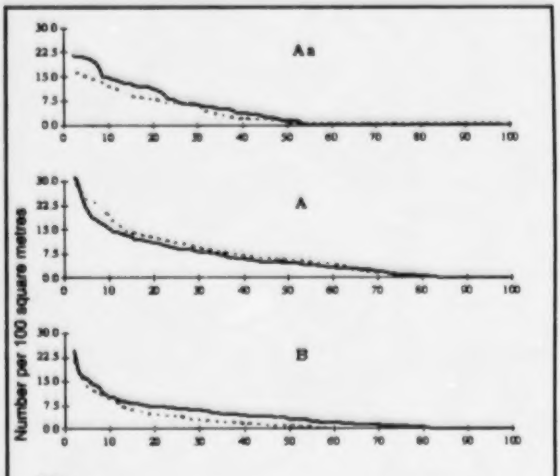

Figure 10. Exceedance curves, by channel type, for the abundance (no./100 m²) of overyearling cutthroat trout at stations in roaded (dashed lines) and unroaded (solid lines) landscapes, Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95. Sample sizes were 71 Aa-, 174 A-, 158 B-, and 136 C-type stations in roaded landscapes, and 92 Aa-, 185 A-, 129 B-, and 53 C-type stations in unroaded landscapes.

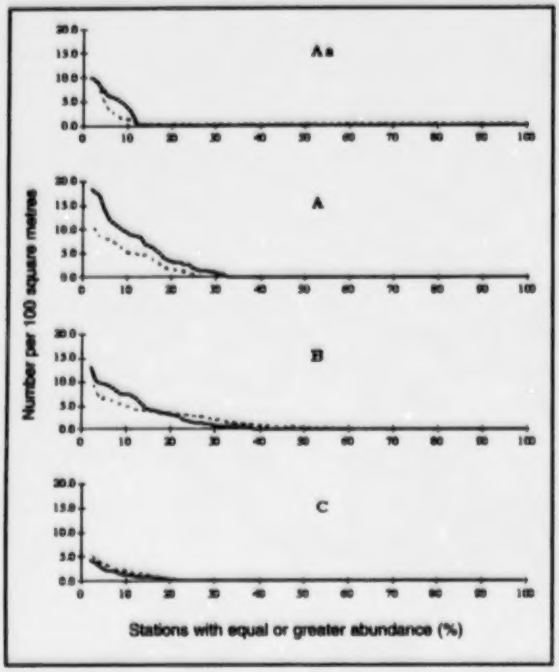

Figure 11. Exceedance curves, by channel type, for the abundance (no./100 m²) of overyearling rainbow-steelhead at stations in roaded (dashed lines) and unroaded (solid lines) landscapes, Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95. Sample sizes were 71 Aa-; 174 A-, 158 B, and 136 C-type stations in roaded landscapes, and 92 Aa-, 185 A-, 129 B-, and 53 C-type stations in unroaded landscapes.

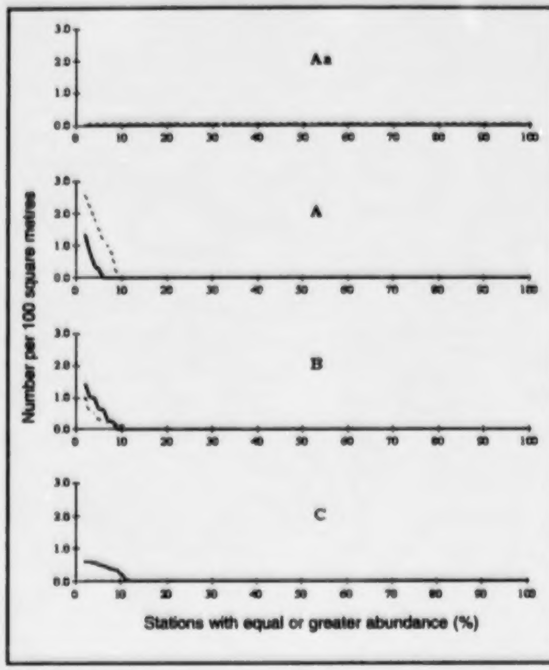

Figure 12. Exceedance curves, by channel type, for the abundance (no./100 m²) of overyearling bull trout at stations in roaded (dashed lines) and unroaded (solid lines) landscapes, Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95. Sample sizes were 71 Aa-, 174 A-, 158 B-, and 136 C-type stations in roaded landscapes, and 92 Aa-, 185 A-, 129 B-, and 53 C-type stations in unroaded landscapes.

Discussion

Differences between streams in roaded and unroaded areas of the CNF reflect interactions between uneven patterns of anthropogenic disturbance and natural variation across the landscape. Survey data suggest that most of the highest quality fish habitat in the CNF is in unroaded areas, where levels of fine streambed sediment are lower and pool abundance tends to be slightly higher than in roaded landscapes. These landscape-level differences in stream features exist in spite of an extensive history of catastrophic forest fires in many areas that lack roads. Given that a high proportion of the CNF's unroaded landscapes are still slowly recovering from stand-replacing fires that occurred in the early 1900s, anthropogenic impacts on streams in the roaded landscapes may have been even greater than suggested by current differences between roaded and unroaded streams.

The substantial difference in sediment levels between roaded and unroaded streams on the CNF is not surprising given the predominance of granitic batholith geology in the area. Arnold and Lundeen (1968) attributed 91% of the annual sediment production in batholithic geology within the South Fork Salmon River, Idaho to roads and skid trails, at a time when about 14% of that river's basin had been developed (D. Burns, Payette National Forest, McCall, ID, personal communication). Most of this increase was associated with logging roads that produced sediment at 770 times the undisturbed rate (Megahan and Kidd 1972). Burns (1984) found significantly higher streambed embeddedness in managed (roaded and logged) versus unroaded watersheds in that same river basin.

The slightly higher abundance of pool habitat observed in unroaded versus roaded CNF streams is similar to findings by Overton et al. (1993) and Cross

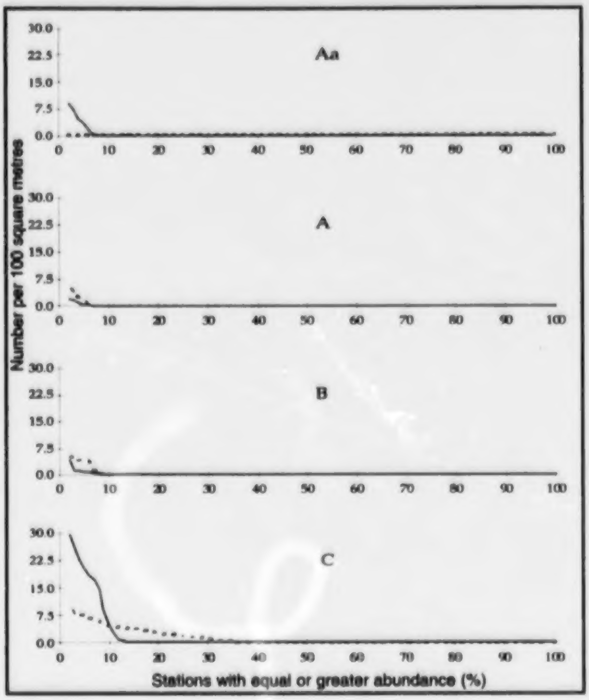

Figure 13. Exceedance curves, by channel type, for the abundance (no./100 m²) of overyearling brook trout at stations in roaded (dashed lines) and unroaded (solid lines) landscapes, Clearwater National Forest, Clearwater River sub-basin, Idaho, 1989-95. Sample sizes were 71 Aa-, 174 A-, 158 B-, and 136 C-type stations in roaded landscapes, and 92 Aa-, 185 A-, 129 B-, and 53 C-type stations in unroaded landscapes.

and Everest (1995) on other national forests in Idaho, and by Ralph et al. (1994) on streams in Western Washington. It is also consistent with McIntosh et al.'s (1994) finding of pool loss in most managed (i.e., roaded) interior CRB streams examined during repeats of surveys completed 50 years earlier. More recent work by McIntosh (1995) suggests that increases in pool abundance observed in this region over the last 50 years have generally been restricted to unroaded watersheds or drainage networks downstream of large unroaded watersheds. Some of those increases were undoubtedly associated with stream recovery from the impacts of catastrophic fires like those experienced across many of the CNF's unroaded areas.

There was significantly more coarse woody debris along A- and Aa-type channels in roaded areas than in unroaded areas even though riparian timber harvest has removed some of the potential debris from areas along many streams within the CNF. I suspect that the greater quantities of coarse woody debris along many of these roaded streams are related to the effects that historic wildfires had on riparian conifers in many currently unroaded areas, and a higher abundance of timber within currently roaded areas prior to development. There was significantly less acting and potential woody debris along C-type channels in roaded watersheds than in unroaded areas. Although this difference may reflect anthropogenic impacts, it could also be the result of past fires and ongoing natural recovery processes in many unroaded watersheds.

Westslope cutthroat, the most common trout in the CNF, tended to be more abundant within unroaded than in roaded landscapes, particularly in lower gradient B- and C-type channels where the most pronounced cumulative effects of past disturbances might be expected. While observed between-landscape differences in habitat quality and fragmentation offer a reasonable explanation for the generally lower abundance of cutthroat trout in these lower gradient channels, differential angling mortality may also be a factor. This is because roads improve angler access and westslope cutthroat trout (MacPhee 1966; Lewynsky 1986) and other species native to the CNF are vulnerable to angling. Regardless of the relative role of angling, differences in the abundance of cutthroat trout between the CNF's roaded and unroaded landscapes seem largely attributable to differences in levels of human activity. Although some human activities are considerably more invasive than others, most are either enabled or expanded by the construction of roads (Henjum et al. 1994; Noss and Cooperrider 1994).

Bull trout are less abundant and more narrowly distributed in the CNF than are westslope cutthroat trout. Known bull trout spawning areas, as identified by the presence of young fish, are rare and about equally represented in roaded CNF watersheds. This might seem contrary to the idea that bull trout do best in pristine areas or those least affected by anthropogenic disturbance (Ratliff and Howell 1992; Rieman and McIntyre 1993). However, I suspect that the current spatial distribution of spawning areas for bull trout within the CNF reflects a combination of very specific habitat requirements and a legacy of elevated stream temperatures in many unroaded watersheds following the catastrophic fires which occurred in the early 1900s. Regardless of where they find suitable spawning and early rearing habitat, most fluvial

bull trout within the CNF still have the opportunity to spend much of adulthood in high quality rivers that drain large unroaded areas.

Many land managers in the CRB are developing new conservation strategies in an effort to maintain or restore populations of aquatic species currently in decline. One element of many such strategies is the protection of strong populations in refuge areas, particularly in the near term. Data collected on the CNF suggest that large unroaded areas with a relatively recent (<100 yr) history of catastrophic, stand-replacing wildfires can provide suitable refugia for native trout populations. Such areas also provide opportunities for maintaining naturally functioning C-type channels that may be critical to certain aquatic species. However, although these landscapes provide abundant refugia for widely distributed species like cutthroat trout, they may give less adequate refuge to bull trout because key spawning and juvenile-rearing areas for this species appear to be rare and frequently occur in roaded watersheds. Conservation of aquatic species whose critical habitats are damaged or threatened will require strategies like those proposed by Frissell (1993), Henjum et al. (1994), Reeves et al. (1995), and others, which would both protect remaining high-quality habitats and lead to the development of new high-quality habitat over time.

There is currently much talk in the CRB and elsewhere in the US about the need for ecosystem management (EM) on national forests and other lands. Some individuals advocating EM seem intent on applying it first to unroaded public landscapes. Stream conditions within the CNF provide an important backdrop for discussions of EM and of how best to manage unroaded public lands in the U.S. Current understanding of landscape-level patterns of disturbance, recovery, and variation in streams is rudimentary. It remains to be seen whether we can begin actively managing sensitive unroaded landscapes, constructing new roads and dramatically expanding the level of human activity within them, while maintaining the kinds of high-quality aquatic habitat and strong populations of native trout they currently sustain. As suggested by Rhodes et al. (1994), it would seem prudent to gain confidence in our ability to create new refugia for sensitive aquatic species before placing at risk those that already exist.

Acknowledgments

I would like to acknowledge several contributions. The CNF funded collection of all data reported here and has shown a stronger commitment to assembling a good stream database than any other land management unit of which I am aware. Responsibility for this commitment rests with CNF resource specialists, particularly Al Espinosa and Pat Murphy. I was more than greatly assisted in collection of the data by more than a dozen Clearwater BioStudies, Inc. biologists and technicians. Many of those individuals dedicated entire summers to carefully examining streams and sampling fish populations under difficult field conditions in remote and rugged terrain. Early compilations and preliminary analyses of most of the data reported here were funded by the U.S. Forest Service's Eastside Ecosystem Management Project. Comments received from several anonymous reviewers helped to substantially improve an earlier version of this manuscript.

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Cumulative Effects Analysis of Land-Use in the Carbondale River Catchment: Implications for Fish Management

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Abstract

Recent studies analyzing the cumulative effects of land-use in forested ecosystems show promise for using geographic information systems (GIS) as simple tools for making useful cumulative-effects assessments suitable for routine management applications. We demonstrate that a particular GIS-based watershed analysis approach recently developed for routine use in the interior of adjacent British Columbia, the Interior Watershed Assessment Procedure (IWAP), is applicable to the analysis of cumulative effects of land-use on Alberta's Eastern Slopes, providing results of immediate utility for managing land-uses and fish habitat. We applied a slight adaptation of the IWAP in order to quantitatively document and analyze land-use in the Carbondale River watershed, a representative small river basin on the Eastern Slopes, using existing publicly-available digital geographic data for the area. The results predicted that the basin as a whole, and all of its sub-basins, are at very high risk of damage from increased peak flows, increased surface erosion, or the interaction of increased peak flows and increased surface erosion, resulting from land-use in the basin. The riparian zones of the entire basin and all but one of the sub-basins are predicted to be at high risk of damage as the result of extensive logging of streambanks. These assessments of risk were consistent with previously reported independent observations of extensive damage to stream channels and riparian habitat in the field, with one exception: the procedure failed to predict extensive streambank damage that had previously been independently observed in Gardiner Creek. Delivery of water and sediment are major factors shaping the basin's mainly gravelbed channels, and these basin-wide factors have been fundamentally changed from the natural condition by past and current land-use. The results of this study imply that land-use patterns in the basin must be improved to significantly improve fish habitat in the long term. Accordingly, all roads not required for basic access should be decommissioned and equivalent clearcut area should be reduced to restore the ecological integrity of the Carbondale basin.

Sawyer, M.D. and Mayhood, D.W. 1998. Cumulative Effects Analysis of Land-Use in the Carbondale River Catchment: Implications for Fish Management. Pages 429–444 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

The Rocky Mountain region in Canada is under heavy development pressure, particularly for resource extraction, tourism, recreation, grazing and, locally, from intensive urbanization. The Eastern Slopes in Alberta, a 108 000-km^2 area of mountains, foothills, forests and grasslands east of the Continental Divide, is no exception. The 90 000-km^2 portion of the region under Alberta Government jurisdiction is managed under a multiple-use policy that explicitly promotes unlimited growth in resource development while at the same time proposing to maintain key watershed and recreational values (Alberta Energy and Natural Resources 1984). Nevertheless, natural resources in the Eastern Slopes are to be developed, managed and protected "in a manner consistent with principles of conservation and environmental protection" (Alberta Energy and Natural Resources 1984).

Even though these management goals may be met initially when development intensity is low, it is obvious that they must ultimately be incompatible when unstinted development overwhelms natural systems. We endorse an ecosystem-based management approach (e.g., Agee and Johnson 1988; Keiter and Boyce 1991; FEMAT 1993; AFSEEE 1995) as a realistic replacement for this and similar policies. We propose that the goal of ecosystem management in the Rocky Mountains be to accommodate a level of development that maintains natural ecosystems intact in perpetuity. To meet this goal, an evaluation of the existing condition of the landscape relative to its natural condition — a cumulative-effects analysis — is needed as a basis for setting limits to culturally induced change.

We understand cumulative effects to be the total accumulated changes induced by humans in the environment. Individually these effects may be minor, but collectively they can be significant. It is the cumulative effects of all human-induced environmental change that determine the limits within which we can modify ecosystems without destroying their value to us and the essential services they provide. For this reason, we must continually assess the cumulative effects of all human activities in a region if we wish to manage and sustain its renewable resources.

Current views of cumulative-effects assessment see the process as poorly understood, difficult and complex, with few agreed-upon procedures and methods suitable for general application (e.g., numerous papers in Kennedy 1994; Hegmann and Yarranton 1995). Indeed, many of the assessments that have been done have been extensive, tedious, expensive and time-consuming (e.g., Leathe and Enk 1985; Antoniuk 1994; Okrainetz 1994; Smith 1994; Duinker 1994; and Hegmann and Yarranton 1995 describe many other examples). Finally, there is the problem of what to do when the cumulative effects have been identified and quantified. What degree of change is allowed? Some imply that these limits are unreasonably difficult to ascertain (Hegmann and Yarranton 1995). Cumulative effects of many development proposals often are not seriously assessed for these and less savory reasons, even when it is required by policy or law (Nikiforuk 1997).

In our view the many perceived difficulties in assessing cumulative effects are often of little practical importance. For most purposes it is seldom necessary to evaluate every potential interaction for its cumulative effect. Some simple and quick methods are available that, when combined with existing guidelines and standards, can produce results of immediate value for making environmental management decisions.

For example, several recent studies have used computerized geographic information systems (GIS) to assist in the spatial analysis of the cumulative effects of land-use on ecosystems (e.g., Forest Ecosystem Management Recovery Team 1993; Case et al. 1994; several studies cited by Spaling and Smit 1994). These tools are quantitative, fast, cheap and make use of existing, frequently updated, digital map data. GIS-based procedures rank among the better approaches in meeting the objectives of cumulative-effects assessment (Spaling and Smit 1994). At the same time, quantitative or semi-quantitative standards for human disturbance in ecosystems are being developed by some jurisdictions (e.g., B.C. Forest Service 1995a, 1995b). Many important human disturbances in a region can be quantified using a GIS and analyzed according to the existing standards to arrive at defensible decisions about the amount of additional disturbance that can be permitted.

In the study reported here, we demonstrate that a particular GIS-based watershed analysis approach recently developed for the interior of adjacent British Columbia (B.C. Forest Service 1995a) is practical for routine use, and is applicable to the analysis of cumulative effects on Alberta's Eastern Slopes. To do this, we apply the approach to quantitatively document and analyze land-use in a representative small river basin on the Eastern Slopes using existing publicly available digital geographic data for the area. We discuss the applicability of the approach to Alberta

based on broad similarities and differences between the characteristics of the study area and the region for which it was specifically designed. We evaluate the potential impacts of land-use on streams in the study basin according to the published standard criteria of the approach, and compare these assessments to independent observations on land-use effects made previously in fish habitat surveys in the same basin. Finally, we comment on the implications of the findings for managing fish habitat in the study basin.

Study Area

We chose the Carbondale River basin of southwestern Alberta for this demonstration because it is subject to most of the land-uses common on the Eastern Slopes, including logging, mining, grazing, petroleum exploration and development, recreation and off-road vehicle travel (Gibbard and Sheppard 1992). Streams in this basin are regionally important angling waters. The Carbondale River basin is also of considerable interest as a region of high biodiversity, containing many rare species of plants and animals (e.g., Wallis 1994; Alberta Environmental Protection 1995; Gerrand and Sheppard 1995), including two species of fish considered to be at-risk: westslope cutthroat trout and bull trout (Gerrand and Sheppard 1995). The results of a cumulative-effects study in this basin therefore are of broad and immediate practical interest.

Effects of land-use on watersheds are likely to depend on relief, geology, soils, climate and forest cover (B.C. Forest Service 1995a). The significance of these effects on fish will depend on the species present and on their use of the habitat (Hicks et al. 1991).

The Carbondale River is a major tributary of the Castle River in the Oldman River drainage, a part of the South Saskatchewan River system (Fig. 1). The 309-km^2 basin lies in the Front Ranges of the southern Canadian Rocky Mountains on the eastern slopes of the Continental Divide. The basin is topographically rugged, with relief of over 1229 m, from over 2500 m on the divide to 1271 m at the confluence with the Castle River (Fitch 1980a). Bedrock is dominated by shale, sandstone conglomerate and coal of late Triassic to Tertiary age (Gadd 1995). Surficial deposits are mostly moderately to slightly leached silty-sand tills of varying texture, ranging from fine to extremely coarse and silty sand colluvium. Coarse alluvium is associated with stream channels (Bayrock and Reimchen 1980).

Soils, mapped in detail only in the eastern part of the study area, have been described by Twardy and Sehn (1994). Orthic gray luvisols are common, occurring primarily under forests on loam-textured tills, on slopes ranging from 2 to 45%. Water erosion hazard rating of the luvisols is moderate on 5 to 9% slopes, and high on slopes greater than 9%. Orthic dark gray chernozems, particularly common in the eastern portions of the study area under grasslands (Achuff 1992), occur on silty to sandy loam, or on loam- to clay loam-textured till or fluvial veneer over till, on terrain that is gently undulating to moderately rolling (1 to 15% slopes). Water erosion hazard rating of the chernozems is moderate on 9 to 15% slopes and high on slopes greater than 15%. Orthic and lithic regosols, very common on colluvial slopes and on active fluvial landforms, occur on sandy loam to loam-textured weathered bedrock and rock on

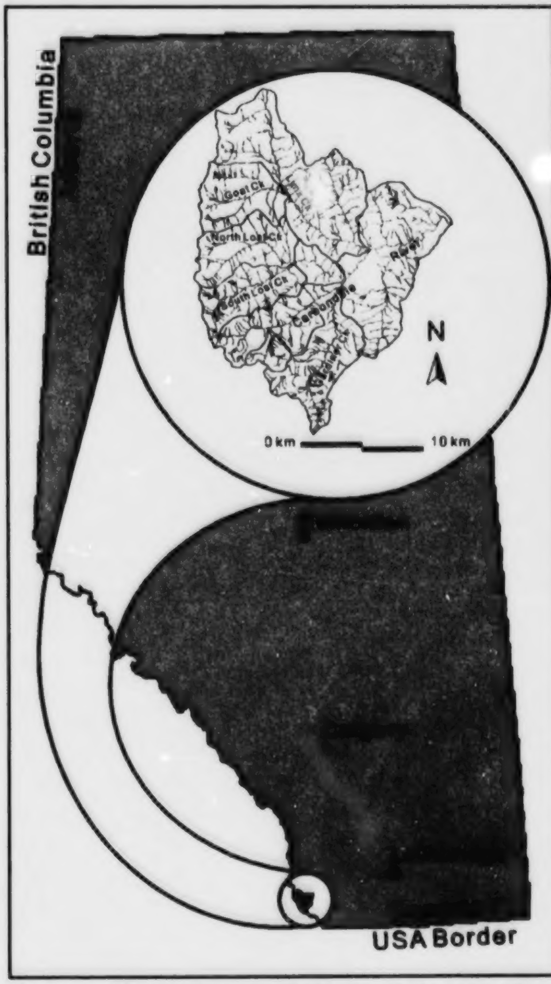

Figure 1. The study watershed: the Carbondale River basin, Alberta, Canada.

strongly to extremely rolling terrain (16 to 70% slopes). Water erosion hazard of regosols is moderate on 9 to 15% slopes and high on slopes greater than 15%

The climate is northern continental, with cold winters and short, cool summers. Mean daily temperatures in January range from -7 to -9°C, and in July are about 16°C (Longley 1972). On average, only 149 days are above freezing (Gadd 1995). The rate of precipitation is among the highest on the Alberta Eastern Slopes: mean annual precipitation ranges from 51 to 107 cm, approximately 30 to 53% of which falls as snow (Longley 1972; Atmospheric Environment Service data for Castle Ranger Station cited by Gadd 1995). The regional snowfall ranges from 221 to 459 cm. Warm, dry and strong chinook winds characteristic of the Eastern Slopes exert a powerful drying influence throughout the year, and are especially prevalent in the study area. These winds are capable of removing most of the snowpack in a matter of days, especially on west-facing slopes. On average, chinooks occur 30 out of 120 winter days (Gadd 1995) but may not occur at all in about one winter out of 10 (Longley 1972).

We did not find streamflow data for the Carbondale River basin, but the hydrology can be expected to display characteristics similar to other Eastern Slopes basins. Streamflows here are highest during spring snowmelt, peaking in late May to early June. Rain-on-snow events during this period have produced large floods, including two greater than 100-year events in the last two decades in the Castle basin. Low flows occur in late winter. Most streams are ice-covered from December through February except during prolonged chinooks, or in areas influenced by groundwater discharge.

The montane sub-region, comprising most of the eastern portion of the study area, is characterized by a savannah of *Pseudotsuga menziesii* (Douglas-fir), *Pinus flexilis* (limber pine) and *Pinus contorta* (lodgepole pine). Subalpine forests dominating the central and western portions vary primarily according to altitude and fire history. Lower subalpine sites disturbed by fire are dominated by lodgepole pine, while *Picea engelmannii* (Engelmann spruce) and *Abies lasiocarpa* (subalpine fir) predominate on cooler, mesic sites not recently burned. At higher elevations *Pinus albicaulis* (whitebark pine) and *Larix lyallii* (subalpine larch) are diagnostic of the transition to the treeless alpine sub-region (Achuff 1992).

Westslope cutthroat trout (*Oncorhynchus clarki lewisi*) and bull trout (*Salvelinus confluentus*) are the only widespread native fishes in the basin. Mountain whitefish (*Prosopium williamsoni*), longnose suckers (*Catostomus catostomus*) and longnose dace (*Rhinichthys cataractae*) are native in the lower reach of the Carbondale River below a waterfall barrier. Rainbow trout (*O. mykiss*) have been widely introduced within the drainage, and appear to have introgressively hybridized with the cutthroats (Fitch 1980a–g, D. Mayhood unpublished data).

Methods

We adapted slightly the B.C. Forest Service's Level I Interior Watershed Assessment Procedure (IWAP) to assess the cumulative effects of existing human disturbance in the Carbondale watershed. The IWAP was developed for use in the interior forested watersheds of British Columbia, including the west slopes of the Rocky Mountains immediately adjacent to the study area (B.C. Forest Service 1995a). The procedure is intended to help land managers understand the type and extent of water-related problems in a basin and to recognize the possible hydrologic implications of proposed future developments in that basin.

The IWAP assesses potential for certain hydrological impacts in a watershed, specifically the potential for (1) changes in peak flows, (2) accelerated surface erosion, (3) changes to riparian zones, and (4) mass wasting, from the 13 indicators of cultural disturbance listed below.

Peak Flow Indicators

1. peak flow index
2. road density above the H_{60} line (elevation above which 60% of the watershed area lies)
3. road density in the entire basin

Surface erosion indicators

4. road density on erodible soils
5. density of roads within 100 m of a watercourse
6. road density on erodible soils within 100 m of a watercourse
7. stream crossing density
8. road density in the entire basin

Riparian buffer indicators

9. proportion of watercourse banks logged
10. proportion of banks of fish-bearing watercourse logged

Mass-wasting

11. landslide density in the watershed

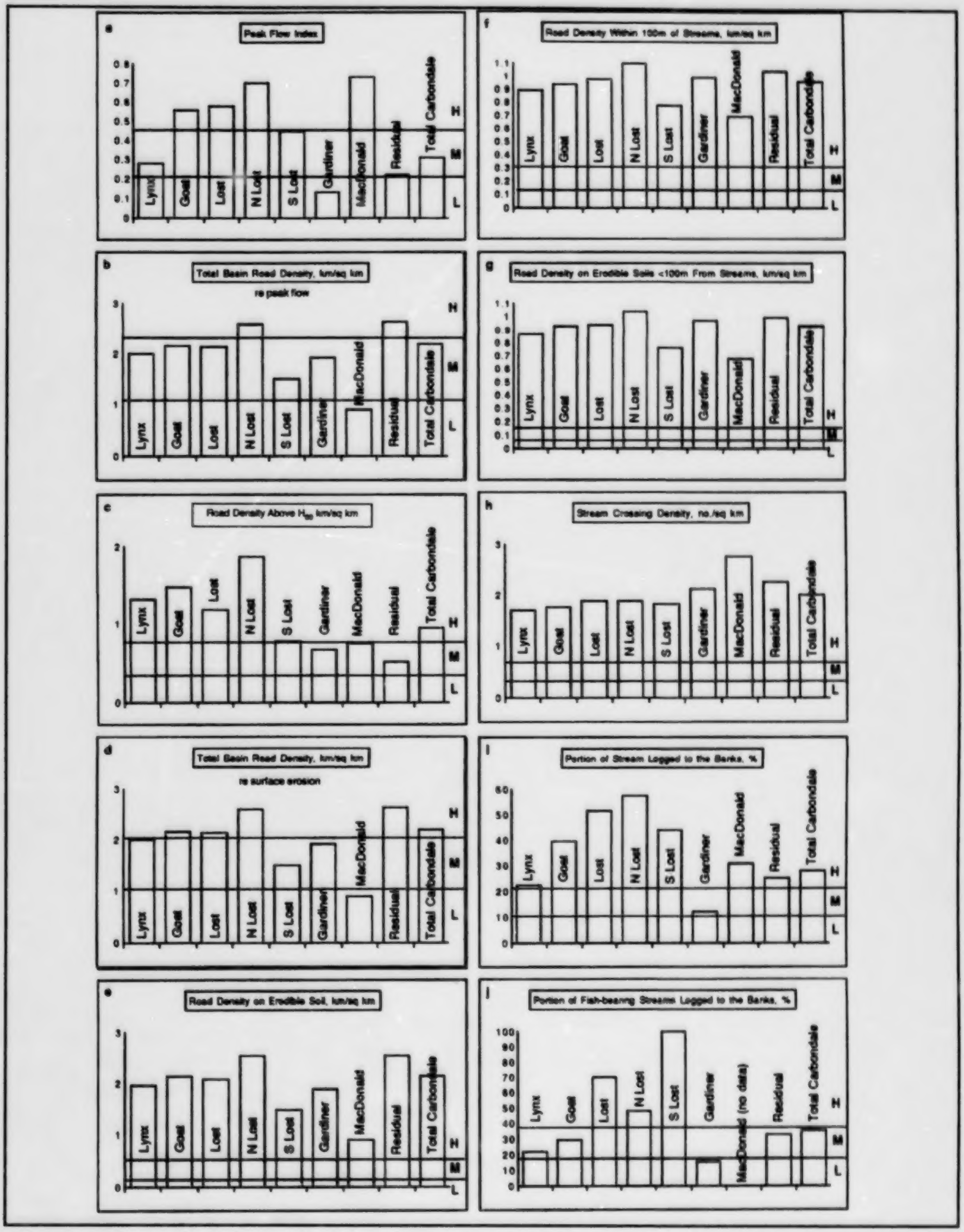

Figure 2. Indicators of environmental impact produced by the IWAP analysis, Carbondale River basin and its sub-basins. H, high; M, moderate; L, low potential impact.

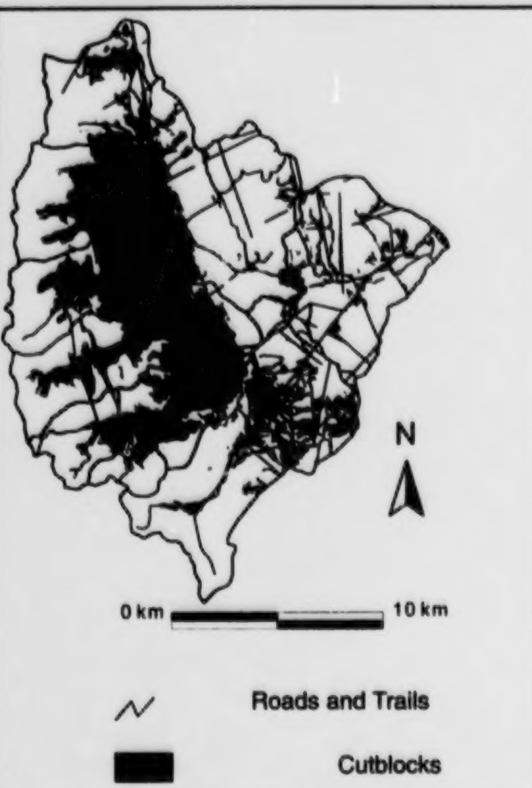

Figure 3. Principal human disturbances from land-use in the Carbondale River basin.

12. road density on unstable or potentially unstable slopes
13. proportion of watercourses logged on slopes exceeding 60%

These indicators are standardized to values between 0 and 1 according to results of studies on 40 watersheds in interior B.C. The indicators in each category are evaluated together to arrive at a hazard index score for that category, then the hazard indices are interpreted in several pairwise matrices to assess the potential for environmental impact arising from their interactions. We undertook to assess peak flow, surface erosion and riparian buffers in this reconnaissance-level study, but did not have the resources to do the landslide counts and acquire the terrain stability data required to make an assessment of mass-wasting potential. Omitting the mass-wasting assessment will tend to underestimate the cumulative effects of human development on the study area.

All digital data necessary for the analysis were purchased from Alberta Environmental Protection and were not modified for this study. Linear data (roads, trails, streams, lakes) and digital elevation models (DEMs) were obtained from 1:20 000 provincial digital base map files. Vegetation data, including species composition and stand disturbance modifiers, were derived from digital Alberta Vegetation Inventory (AVI) files. The data sets are accurate to ±5 m. The linear and DEM data are current to 1994 and vegetation data are current to mid-1995. The data were managed, analyzed and displayed with ArcInfo 3.4.2 (Environmental Systems Research Institute 1995), a personal computer-based GIS. The Unix version of ArcInfo 7.0 was used for some complex analyses not available in version 3.4.2.

We used the calculation and analysis methods detailed in the IWAP Guidebook (B.C. Forest Service 1995a), with only the modifications outlined below. It is critical to be aware that, under the IWAP calculating methods, all density figures are standardized to the total area of the basin or sub-basin in question, even those related to some restricted area. For example, road density within 100 m of a stream is calculated as the length of road within 100 m of a stream divided by the total area of the basin, not just by the area of the 100-m buffer. The resulting figure therefore is not the actual road density within the buffer,

Table 1. Standard IWAP (B.C.) hydrological recovery criteria for partially cut areas, and the Alberta equivalents used in the calculation of equivalent clearcut area in this study

	100% recovery	50% recovery	0% recovery
IWAP	<30% basal area removed	30–60% basal area removed	>60% basal area removed
Alberta	1–25% loss of crown closure	26–75% loss of crown closure	76–100% loss of crown closure

but the overall density of road in that category in the basin in question. The true densities within the restricted areas thus will always be higher—usually much higher—than the figures calculated according to the IWAP.

Equivalent clearcut area (ECA) is used in the IWAP to calculate a peak flow index, a ratio of ECA to total basin area, weighted to account for the greater influence on peak streamflows of clearcuts at higher elevations. The method of calculating ECA had to be slightly modified to accommodate the Alberta data. ECA is the area that has been clearcut, reduced by a factor to account for the hydrological recovery due to forest regeneration (B.C. Forest Service 1995a). The hydrological recovery factor was obtained from the AVI data, which provide the height of regeneration and the loss of crown closure in each logged polygon. Because the IWAP recovery criteria for partially cut sites are based on three categories of basal area removal, whereas the AVI data provided a five-category classification of loss of crown closure, we adjusted the IWAP criteria to suit our data (Table 1). Because restocking in our study area was very low, this adjustment tends to underestimate ECA (i.e., shows greater recovery) relative to that derived from B.C.'s three-category data. Our ECA estimates are therefore somewhat conservative.

Trails for seismic exploration were considered roads in our analysis. Virtually all seismic trails in the study area are used—often heavily—by recreational off-road vehicles (M. Sawyer and M. Judd, unpublished data). Our procedure still underestimates the extent of road and trail development, because recreational off-roaders continually create their own trails throughout the basin (Gibbard and Sheppard 1992; Sheppard 1995; M. Sawyer unpublished data), and these do not necessarily show up in the digital database.

The IWAP uses various calculated watershed indicators (primary indices) to establish hazard index scores (secondary indices) used in bivariate matrices, termed interaction matrices, to create interaction matrix values (tertiary indices). It is these highest-order indices that are used to make recommendations about watershed management. We were able to calculate an interpretation matrix only for the interaction of peak flow and surface erosion. This was a Level 1 analysis, so we did not have the necessary channel stability data collected in a Level 2 analysis (B.C. Forest Service 1996) to calculate values for the three interpretation matrices requiring those data, and as noted above, we did not have the mass-wasting data that would have allowed us to calculate an interpretation matrix for mass-wasting versus peak flow. Instead, we initially rated potential for watershed damage directly from the primary indices in the "watershed report card" (B.C. Forest Service 1995a). We used the ranking scheme of the relative impact guidelines and the scoring conversion table in the IWAP document, which is based on results from 40 test watersheds representing the four interior forest regions in British Columbia (B.C. Forest Service 1995a). We also rated the secondary indices (hazard indices) for the peak flow, surface erosion and riparian buffers impact categories according to the low-medium-high hazard criteria established in the IWAP (B.C. Forest Service 1995).

Soils and their susceptibility to water erosion have been mapped only for the eastern portion of the study area (Twardy and Sehn 1994). Because soil types are closely correlated with surficial geology and the surficial deposits of the study area have been mapped (Bayrock and Reimchen 1980), we identified soil types based on the known distribution of surficial deposits. The erodibility of each soil type is strongly related to slope (Twardy and Sehn 1994). We therefore mapped slope classes using the DEM, and used these together with the map of surficial deposits to map soil erodibility for those areas where site-specific erodibility data were not available.

Results

The IWAP analysis produced a total of 10 indicators of the impact of human disturbance on peak streamflows, surface erosion and riparian buffers. These indicators are summarized in Figure 2 together with the levels of potential low, moderate and high impact. The GIS data from which the ten indicators were calculated are presented in Table 2. Figure 3 maps the total amount of human-induced surface disturbance in the Carbondale basin. Whenever possible, the results for larger basins incorporated those for all included sub-basins. The Goat Creek drainage is a sub-basin of Lynx Creek, and North and South Lost creeks are sub-basins of Lost Creek. MacDonald, Gardiner, Lost and Lynx Creeks are direct tributaries of the Carbondale River.

The sub-basins not included in any of the named basins in the tables or figures were grouped and referred to as the residual Carbondale sub-basin (cf. B.C. Forest Service 1995a).

Peak Flows

Peak flow index. The peak flow index is intended as a measure of the sensitivity of a basin to increases in peak flows resulting from clearcutting. Higher values indicate a greater sensitivity to increased peak flows. The index is calculated as a weighted measure of the proportion of the basin that has been clearcut, the weighting depending on the fraction of clearcutting in a particularly sensitive zone, the upper 60% of the basin that is still snow-covered at the time that streamflows begin to rise in the spring. The elevation of this snowline (the H_{60} line) in the Carbondale basin was calculated from the DEM to be 1622 m.

The peak flow index for the Carbondale basin suggests a moderate potential for increased peak flows from clearcutting overall, but the index for five of the sub-basins was high, and could be described as extremely high in two (Fig. 2a). Two sub-basins had moderate potential for increased peak flows from clearcutting, and only one had a low potential.

Road densities and peak flows. The potential for increased peak flows also increases as the road density increases, because roads act in part as an extension of the surface drainage network, effectively increasing drainage efficiency. The effect is likely to be especially pronounced above the H_{60} line, where most of the meltwater contributing to spring peaks in flows originates.

The potential of overall road densities to increase peak flows was generally ranked moderate in most sub-basins and in the total Carbondale basin, but was rated as high in North Lost Creek and in the residual Carbondale basin (Fig. 2b). In contrast, the potential for road densities above the H_{60} line to increase peak flows could be described as extremely high in four sub-basins, high in one, and moderate in three (Fig. 2c). The potential for increased peak flows from road development above the H_{60} line was rated as high for the Carbondale basin overall.

Surface Erosion

Based on its potential for increasing surface erosion, the overall road density was considered to be high in the Carbondale basin and in several of the sub-basins (Fig. 2d). Potential surface erosion due to roads as measured by all other indicators could reasonably be rated extremely high for every sub-basin (with the exception of road density on erodible soils in the MacDonald sub-basin) and for the Carbondale watershed as a whole (Figs. 2e–h). The two indicators incorporating soil erodibility are extremely high in part because 97% of the Carbondale drainage had erosion hazard ratings of moderate (17%) or high (80%) based on slope, surficial geology and soil type.

Riparian Buffers

Overall, 28% of the total length of watercourse in the Carbondale basin has been logged on at least one bank, leaving little or no buffer strip; 40 to 58% of total watercourse length was logged to at least one bank in four sub-basins (Table 2). The potential for resulting riparian damage was rated high to extreme in the Carbondale watershed and all but one of its sub-basins (Fig. 2i).

Thirty-five percent of the known fish-bearing stream length in the Carbondale basin was logged to the banks on at least one side; this proportion ranged to as high as 70% in the Lost Creek sub-basin and 100% in the South Lost Creek sub-basin (Table 2). Nevertheless, the values overall suggest only a moderate potential impact from logging on streambank fish habitat in the Carbondale basin as a whole, even though the potential impact in the Lost Creek sub-basin is high to extreme (Fig. 2j).

Hazard Indices

Hazard indices summarize the overall potential for impact from the two to five indicators in each impact category. Hazard indices for surface erosion ranked uniformly high throughout the basin (Table 3). Hazard indices for peak flows rated high for the Carbondale basin overall, and for all but two of the sub-basins. Riparian buffer hazard indices were high for the Carbondale basin and all but one of its sub-basins, being low in Gardiner Creek.

Peak Flow–Surface Erosion Interaction

We created an IWAP interaction matrix for the two indicator classes for which we had sufficient data to do so: surface erosion and peak flow. Tertiary index values of 4, the highest possible, indicating very high potential for channel damage from erosion and sediment deposition, were obtained for the total Carbondale catchment and all sub-basins (Table 3).

Table 2. GIS-derived data on the Carbondale basin used in the IWAP analysis

Parameter	Lynx Creek	Goat Creek	Lost Creek	North Lost Creek	South Lost Creek	Gardiner Creek	MacDonald Creek	Residual Carbondale Basin	Total Carbondale Basin
Total area of basin (km^2)	103.43	29.35	65.23	29.99	26.26	36.26	6.49	97.68	309.08
Area above H_{60} line (km^2)	74.11	24.36	46.56	25.17	20.19	25.12	6.07	33.42	185.28
Total ECA (km^2)	22.54	11.96	29.56	15.31	9.55	4.29	4.27	18.65	79.31
ECA above H_{60} (km^2)	12.88	8.90	15.99	11.19	4.39	0.96	0.93	4.92	35.67
Road length (km)	207.94	63.56	139.63	78.02	39.73	69.75	5.88	259.03	682.22
Road length above H_{60} (km)	136.17	43.55	77.18	56.08	20.61	24.48	4.84	50.17	292.83
Road length on erodible soils (km)	203.23	63.01	136.03	76.25	39.05	68.25	5.83	248.20	661.53
Road length ≤ 100m from stream (km)	92.15	27.50	63.81	32.98	20.59	36.08	4.45	101.34	297.84
Road length ≤ 100m from stream on erodible soils (km)	90.05	27.26	61.28	31.25	19.96	35.30	4.41	96.56	287.61
Stream crossings	177	52	123	56	48	77	18	222	617
Total stream length (km)	206.44	57.64	141.86	59.77	66.51	105.45	18.49	204.58	676.82
Length of stream logged (km)	46.54	22.95	73.17	34.42	29.24	12.59	5.72	51.81	189.83
Total fish-bearing stream length (km)[a]	35.84	6.49	24.55	10.50	8.23	16.92	?[b]	25.39	102.70
Length of fish-bearing stream logged (km)	7.85	1.92	17.18	5.05	8.23	2.64	?	8.35	36.02

[a] Minimal estimates. Many tributaries undoubtedly hold fish, but have not been adequately surveyed.

[b] Creek holds fish, but the fish-bearing stream length is not known.

Table 3. Hazard indices and interaction matrix scores for three classes of cumulative effects in the Carbondale Basin. Hazard indices, reported to the number of significant figures supported by the data, are interpreted as < 0.5, low; 0.5–0.7, moderate; > 0.7, high potential for watershed damage. Peak flow vs. surface erosion (interaction matrix) values can be interpreted as indicating very high potential for impact from the combined effects of increased peak flows and surface erosion (B.C. Forest Service 1995).

	Lynx Creek	Goat Creek	Lost Creek	North Lost Creek	South Lost Creek	Gardiner Creek	MacDonald Creek River	Residual Carbondale Basin	Total Carbondale
Peak flows	0.73	0.9	1.0	1.0	0.8	0.50	1.0	0.60	0.70
Surface erosion	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Riparian buffers	0.8	1.0	1.0	1.0	1.0	0.4	1.0	0.8	0.9
Peak flow vs. surface erosion	4	4	4	4	4	4	4	4	4

Discussion

Is the IWAP Applicable to the Carbondale Basin?

The climatic, pedological, geological and vegetational conditions of the Carbondale basin are sufficiently similar to the region for which the IWAP was developed to render it applicable to our study area as well. The IWAP model was developed for use in the B.C. interior, including the eastern slopes in northeastern B.C. and the western slopes of the Rocky Mountains immediately adjacent to the Carbondale basin. Based on similarities in climate, soils and vegetation, the Carbondale drainage is in the same ecological region as the Elk and Flathead drainages of B.C., the southern ecological region of the Canadian Rockies (Gadd 1995). There are some differences between the eastern and western parts of

the region, but even greater differences are noted among the southern, central and northern regions of the western slopes. If the IWAP is generally applicable using these criteria to all of the western slopes of the Rockies, and the northern east slopes as well, it is applicable using these criteria to the Carbondale area as well.

Climate is expected to have the largest effect on the results of watershed analyses; that is why a somewhat different procedure has been developed for the rain-dominated watersheds of coastal B.C. (B.C. Forest Service 1995b). There is a substantial overlap in most climatological measures for the western slopes of the Rockies and for the Carbondale area (Table 4). Although the Castle weather station, representative of the Carbondale basin, is at a higher elevation than all but four of 37 stations on the Rocky Mountain west slopes (Gadd 1995), the Carbondale region falls well within the climatological range for which the IWAP procedure was designed. Annual precipitation and annual snowfall are relatively high in the Carbondale area, probably because most weather stations on the west side of the Rockies have been placed at lower elevations there. It is possible, however, that the strong chinook winds of the Carbondale basin may significantly modify the effects of the area's greater snowfall. The effect of chinooks on IWAP results should be considered in future modifications of the procedure for the Rocky Mountain eastern slopes.

Potential Effects on Channels, Riparian Zones and Fish Habitat

Increases in peak flows and surface erosion, and logging in the riparian zone can be expected to affect channel morphology, bank stability, riparian and aquatic ecosystem function, and fish habitat. In alluvial channels, such as those that predominate in the Carbondale basin, the physical effects are broadly predictable from basic principles of stream geomorphology, although the details are often highly complex and unpredictable (Schumm 1977; Rosgen 1978; Heede 1980; Gordon et al. 1992). In general, any change in the supply of water or sediment to alluvial channels can be expected to force the channel to adjust its bed or morphology to accommodate the change.

High values for the peak flow–surface erosion index indicate that there is a high probability of damage to stream channels from increases in peak flow and surface erosion. Higher peak flows will increase the erosive power of the stream and its ability to transport sediment, including suspended particles, bedload and large woody debris. Greater surface erosion contributes more sediment to channels. High riparian hazard potentials suggest that there is a high potential for bank erosion and, in the case of fish-bearing reaches, a high potential that erosion and/or loss of large woody debris, cover and shade will damage fish habitat. Logged streambanks are potentially more erodible, and slash is likely to be dumped in the channel during streamside logging operations (Chamberlin et al. 1991, Swanston 1991).

Results reasonably to be expected from increased peak flows in the alluvial channels of the Carbondale basin are flooding or bank erosion with consequent logjams and channel widening (Rosgen 1978; Heede 1980; Chamberlin et al. 1991). Reasonably expected results of increased surface erosion are in-channel sediment wedge and bar formation and channel infilling, causing streams to flood and erode their banks, creating further logjams (Heede 1980; Chamberlin et al. 1991; Swanston 1991; Gordon et al. 1992). Riparian logging might reasonably be expected to result in increased water temperatures,

Table 4. Number of Rocky Mountain west slope (interior B.C.) weather stations recording higher and lower values than the Castle weather station (Rocky Mountain east slope, Carbondale area) for selected climatological criteria. Tabulated from Environment Canada data summarized by Gadd (1995).

	Higher	Lower	Identical	No. of stations
Elevation, m	4	33	0	37
mean annual temperature	16	19	2	37
frost-free days	23	12	0	35
daily Jan. low temperature	22	15	0	37
daily Jul. high temperature	12	25	0	37
annual precipitation	4	33	0	37
annual snowfall	3	34	0	37
no. of days with rain	22	14	0	36
no. of days with snow	11	26	0	37

increased autotrophic production, short-term increases in fine particulate organic matter, eroded banks and logjam formation from slash left in the channel, with a later longterm reduction in large woody debris contributed to the channel (Chamberlin et al. 1991; Murphy and Meehan 1991). The cumulative result of increases in peak flow, surface erosion and riparian logging will depend on the relative increases in the peak flows and surface erosion, and for riparian logging, on the nature of the streambank soils and the quality of the logging operation (Chamberlin et al. 1991; Swanston 1991). Our analysis suggests that the potential for these kinds of damage is generally high, and often extreme, in the Carbondale basin.

The effects of such damage to the stream channels are likely to be generally unfavourable fish (Hicks et al. 1991). On the positive side, moderate increases in water temperature, autotrophic production and fine particulate organic matter might tend to improve fish abundance or growth in small streams (Murphy and Meehan 1991). Moderate short-term increases in large woody debris also temporarily increase cover for fish, and channel widening may increase shallow-water habitat for young-of-the-year and young juveniles.

These positive factors, however, are likely to be more than offset by many negative consequences. Bank erosion, channel widening and channel infilling reduce deepwater habitat required by late juveniles and adults, especially during the critical overwintering period. Bank erosion and longterm decreases in the supply of large woody debris reduce cover. Increased sedimentation of fine particles reduces the quality of spawning and incubation substrate and fills in interstitial cover for early juveniles (Leathe and Enk 1985). Finally, road crossings of streams, especially where culverts are used, can impede or block movements of fish, especially juveniles. Road crossings also play a role in directing eroded sediments into watercourses (Furniss et al. 1991).

These expected effects of increased peak flows, surface erosion and riparian logging in the Carbondale basin can now be used at least in a qualitative way to help determine whether the IWAP evaluations obtained for the watershed are realistic.

Are the IWAP Hazard Evaluations Realistic?

The hazard assessments in this study are, with few exceptions, consistent with independent observations of erosion, sediment deposition and channel changes previously reported in the Carbondale basin. In particular, the 1979 fish habitat surveys of Fitch (1980a–g) documented pervasive, widespread channel damage in the Carbondale River and its major tributaries that he attributed to logging and road effects. His observations will be discussed and compared to our independent assessment in downstream order, from headwaters to the mainstem, to reflect the direction of influence of watershed disturbance on the channels.

Gardiner Creek. Numerous fords on the road paralleling the creek contributed to sedimentation (Fitch 1980b). Clearcuts on the floodplain near the mouth created unstable streambanks and a wide and meandering channel. Some sections of the stream were damaged by flooding. Fitch (1980b) calculated that 23% of the banks of Gardiner Creek were unstable as a result of logging.

Our IWAP analysis for this sub-basin indicated a high potential for channel damage from surface erosion, and the combined effects of surface erosion and peak flows (Fig. 2, Table 3). It is therefore consistent with the previously observed sedimentation and flooding problems. In contrast, the IWAP hazard index for riparian buffers suggests a low potential for streambank damage, inconsistent with Fitch's observations of considerable streambank damage. We are uncertain of the reason for this inconsistency. It is possible that there has been sufficient recovery in this small basin since it was logged that older bankside cutting no longer affects the riparian buffer calculations. Field observations of current conditions could help to resolve the matter.

South Lost Creek. Logging had a major impact on this creek (Fitch 1980c). Clearcut banks, blowdown, poor road alignment, poor bridge placement and numerous trail crossings contributed to increased bank erosion, sedimentation and flooding. Logjams from logging and blowdown had created some good fish habitat, but increased bank erosion. Fully 30% of the banks of the stream were unstable as a result of logging.

The watershed assessment of this sub-basin in the present study indicated high to very high potential for channel and riparian damage from increased peak flows, surface erosion and bank instability (Fig. 2, Table 3). Many of the effects we would expect from such changes (see section of this paper entitled Potential Effects on Channels, Riparian Zones and Fish Habitat, above) were observed by Fitch (1980c).

North Lost Creek. This drainage basin was extensively logged with a major impact on the stream (Fitch 1980d). Bank erosion was increased along the stream

by clearcuts, road alignment, bridge placement, trail crossings and logjams. Nineteen percent of the banks were unstable as a result of logging.

The present watershed analysis indicated a uniformly high potential for channel and riparian damage from increases in peak flows, surface erosion and streambank logging in this sub-basin (Fig. 2, Table 3). The bank damage observed by Fitch (1980d) is consistent with what could be predicted from these high indices (see section of this paper entitled Potential Effects on Channels, Riparian Zones and Fish Habitat, above).

Lost Creek. Fitch (1980e) drew attention to the influence of the logging haul road that paralleled the creek for most of its length. The road occupied the floodplain and sidehill cuts above the stream, and was a major source of sediment. In fact, 26% of the banks were unstable due to logging roads.

The high hazard values for increases in surface erosion in our watershed analysis of this sub-basin (Table 3) reflects in part the high near-stream road density in the basin (Fig. 2f), and is consistent with Fitch's (1980e) road sediment observations. On the other hand, this indicator does not precisely reflect the bank stability problems associated with roads encroaching on streambanks noted by Fitch (1980e), because it includes all roads within 100 m of the streambanks.

Goat Creek. Clearcuts extended to the edge of the banks near the headwaters, rendering them unstable (Fitch 1980f). Overall, 8% of the banks of the creek were unstable due to logging. Braided channels and unstable banks attributed to flood damage were found near the middle and at the mouth of Goat Creek (Fitch 1980f). More recently, Sheppard (1994) documented several instances of stream damage from logging and roads in the Goat Creek drainage. These included unculverted tributary crossings that diverted watercourses down roads—or cut through them—to deposit silt in Goat Creek; an undersized culvert crossing of Goat Creek that had washed out, depositing considerable silt in the creek; and blowdown depositing trees and silty rootwads in the creek from logging to the streambank.

The watershed analysis of Goat Creek basin in this study indicated high potential for damage from increased peak flows, surface erosion and riparian logging (Fig. 2, Table 3).The channel braiding and unstable banks reported by Fitch (1980f) are consistent with the channel infilling, flooding and bank instability expected from increases in sedimentation arising from increased surface erosion and peak flows, and from streamside logging (see section of this paper entitled Potential Effects on Channels, Riparian Zones and Fish Habitat, above).

*Lynx Creek:*Fitch (1980g) stated that extensive logging in the headwaters and tributaries has had a major influence on this stream. He noted that some sections of the creek have been rerouted to accommodate a road, and the road parallels the stream, in some areas encroaching onto the floodplain. Trout habitat below the Goat Creek confluence has deteriorated due to flooding. Overall in Lynx Creek, 22% of the banks were unstable as a result of logging.

In contrast, Pisces Environmental Services Ltd. (1992) took issue with Fitch's interpretation (1980g) that extensive logging in the headwaters of Lynx Creek has had a major influence on that stream. The company felt that in the absence of historical streamflow data it was not possible to determine if the flood damage to Lynx Creek is related to hydrological changes resulting from clearcut logging. Pisces (1992) further stated that there was little evidence of direct impacts of logging: the streambanks had been left intact adjacent to cutblocks, and the most evident effects are related to increased sediment and turbidity loads caused by surface runoff from roads. Pisces (1992) observed that, for approximately 3 km below Goat Creek, Lynx Creek is unconfined, exhibiting evidence of lateral instability and high bedload movement. Above the Goat Creek confluence, debris jams cause channel changes. Woody debris was abundant in a reach of dense, unlogged mature spruce, and was more common there than elsewhere in Lynx Creek. The company attributed the abundance of woody debris in the stream to logging.

Of concern to us in this study is the cumulative impact of cultural disturbance. In his study of the Lynx Creek watershed, Fitch (1980f,g) systematically collected quantitative data on bank stability along almost the entire length of Lynx Creek and most of its main tributary, Goat Creek, looking specifically for any obvious causes for it. His data convince us that streambank logging has been a significant factor creating bank instability in the basin. There appears to be agreement on the point of channel instability in Lynx Creek below the confluence of Goat Creek. Neither investigator attributed a cause to this, but it is at least not inconsistent with impacts to be expected from several of the logging-related disturbances quantified above (see section of this paper entitled Potential Effects on Channels, Riparian Zones and Fish Habitat, above). The abundant woody debris from logging in Lynx Creek, and of siltation from

roads noted by Pisces (1992), are human disturbances primarily attributable to logging. Finally, Pisces (1992) reported that shallow-water habitats (riffles and runs) comprised 93 percent of the total autumn habitat of a 9.5-km reach of Lynx Creek, while less than one percent of the reach consisted of deep pools, runs and flats. These observations are evidence of channel aggradation and widening that is consistent with the effects to be expected from the logging and road development noted by Fitch (1980f,g; see section of this paper entitled Potential Effects on Channels, Riparian Zones and Fish Habitat, above).

Carbondale River: Fitch (1980a) noted that the percentage of sand-silt substrate increased below Lost and Lynx creeks. Overall, 9% of the banks of the Carbondale River mainstem were unstable as a result of logging.

As noted above, the results of the IWAP indicated high potential for damage from increased peak flows, surface erosion and riparian logging in all sub-basins of the Lynx and Lost Creek watersheds (Table 3). Deposition in the Carbondale mainstem of sediments eroded from upstream sites is a result reasonably to be expected (see section of this paper entitled Potential Effects on Channels, Riparian Zones and Fish Habitat, above); thus our watershed analysis is consistent with Fitch's (1980a) observation of an increased proportion of fines in the mainstem substrates below Lynx and Lost creeks.

In short, the IWAP hazard estimates calculated in this paper are consistent with the extensive channel damage in the Carbondale basin documented by Fitch (1980a–g) and others, and attributable primarily to roads and logging. This is perhaps not surprising, given the intensity and extent of human disturbance in the basin. The principal exception was the failure of the watershed analysis to predict the extensive damage to streambanks on Gardiner Creek previously observed by Fitch (1980b). Additional field observations are needed to resolve this issue.

Appropriate Management

This study provides several concrete examples of how the IWAP can be used to guide management in watersheds.

1. The IWAP document presents specific recommendations for action when hazard indices exceed certain values. In this study, the hazard indices were sufficiently high in various combinations that a more detailed Level 2 watershed analysis (B.C. Forest Service 1996) is recommended for all sub-basins and the Carbondale watershed as a whole (Table 3, cf. B.C. Forest Service 1995a). The Level 2 watershed analysis is a channel assessment involving field and office studies intended to estimate the actual level of channel disturbance associated with land-use practices in the subject basin. Where appropriate, the channel impact scores obtained from the Level 2 analysis are used in the interaction matrices to arrive at specific recommendations for management action, as in (2), below.
2. The IWAP document recommends specific management actions when risk assessments are moderate, high or very high as indicated by the interaction matrix scores. The interaction matrix of peak flow versus surface-erosion hazards for all sub-basins and the Carbondale basin as a whole had extreme risk values of 4 in this analysis. For catchments with interaction values of 4 in the peak flow–surface erosion matrix, B.C. Forest Service (1995) recommends the following responses:
   a. initiate an assessment of sediment sources;
   b. permanently deactivate as many roads as possible, consistent with access requirements;
   c. disallow additional roads in sensitive areas; and
   d. reduce the equivalent clearcut area over the entire watershed.

   Given the high peak flow–surface erosion interaction scores of our analysis (Table 3) and the independent observations of channel degradation previously documented by others in the Carbondale basin (Fitch 1980a–g; Pisces 1992; Sheppard 1994), immediate action on these recommendations is justified. Channel assessments conducted concurrently under (1), above should be used to guide this work. It is especially important for fish management that the amount of road be substantially reduced in this basin. The amount of road is the single most important disturbance measure in the IWAP, undoubtedly because roads have such profound effects on sediment and water delivery to channels, with correspondingly far-reaching impacts on fish habitat (Furniss et al. 1991; Waters 1995; Rieman and Clayton 1997). Several studies have related various measures of the amount of road directly to negative effects on fish and their habitats (Furniss et al. 1991 provide a review; see also Leathe and Enk 1985; Eaglin and Hubert 1993; Myers and Swanson 1995).
3. By using the GIS and IWAP as a simple analytical model together with cost estimates, it is possible

to estimate which combination of restoration approaches would provide maximum restoration (minimize IWAP impact indicators) at the least cost. Roadbeds can be satisfactorily decommissioned for perhaps $5000/km (Harr and Nichols 1993). If the entire 682-km road and seismic trail network of the Carbondale basin were to be restored to a more natural condition, the cost could conceivably be in the order of $3.4 million. If the work focused on selectively decommissioning only the most damaging roads and trails, the cost could be substantially reduced. Decommissioning all of the 288 km of roads and trails on erodible soils within 100 m of a watercourse, for example, would reduce the cost to perhaps $1.4 million.

4. Currently efforts are being made in the Carbondale basin to improve fish habitat with instream structures (Pisces 1992). This study suggests that such an approach is likely to fail, because the channel is not the problem; it simply displays the symptoms of widespread watershed degradation. The watershed analysis presented here strongly suggests that in-stream work will provide only short-term improvement at best. The delivery of water and sediment are major factors shaping the basin's mainly gravelbed channels, and these basin-wide factors have been fundamentally changed from the natural condition by past and current land-use. The results of this study imply that land-use patterns in the basin must be improved to improve fish habitat significantly in the long term.

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Freeze-Core Sampling for Sediment Intrusion from Road Stream Crossings in Alberta's Foothills: A Preliminary Discussion

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Abstract

Sediment intrusion into streambed gravel can impair aquatic habitat and can negatively affect fish populations. Road stream crossings from industrial development are a primary source of sediment for erosion and sedimentation into Alberta's foothills streams. Local information on the extent and effects of sediment intrusion from road stream crossings is limited, and sampling and monitoring methods suitable for local conditions are poorly developed. Such information is needed to evaluate guidelines and operating ground rules and to perform environmental audits. The purpose of this research is to provide baseline information on the level of sediment intrusion in foothill streams of west-central Alberta and to adapt existing freeze-core sampling techniques for local use. The hypothesis selected for testing was, "is the degree of sediment intrusion higher downstream of road stream crossings than upstream?" In this paper, first year observations are reported and used as a basis to discuss the logic and suitability of sampling methods used for sediment intrusion.

Spillios, L. C., and Rothwell, R. L. 1998. Freeze-core sampling for sediment intrusion from road stream crossings in Alberta's foothills: a preliminary discussion. Pages 445–450 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

Background Information

The Government of Alberta has identified bull trout [*Salvelinus confluentus* (Suckley)] as a species of special concern (Alberta Environmental Protection 1994) and protected it by limiting sport fish limits to zero (Alberta Environmental Protection 1997). The extensive network of resource extraction roads in the foothills may have been a significant factor in the decline of this species. Road stream crossings are a primary source of erosion and associated sedimentation in streams (MacDonald et al. 1991). During rainfall and snowmelt, suspended sediment concentrations downstream of crossings can be high (Rothwell 1983) and can be harmful to fish (MacDonald et al. 1991). Such concentrations can be short-lived, which makes monitoring logistically difficult and often expensive. Following high flows, suspended sediment settles on and into streambed gravels, which can impair aquatic habitat and result in the mortality of incubating fish (Sterling 1992). Sediment intrusion can affect fish populations by suffocating fish eggs, hindering the removal of metabolites, and preventing newly hatched fish from emerging (MacDonald et al. 1991). It can also disturb benthic macro-invertebrate populations, which inhabit the interstitial spaces of streambed gravel (MacDonald et al. 1991).

Sediment intrusion is the focus of this study on the basis that it may result in long-term habitat alteration and may have more permanent effects on fish habitat and community structure than suspended sediment loads. Resource industries of Alberta, such as forestry and petroleum, must follow operating rules designed to minimize erosion and stream sedimentation (Alberta Environmental Protection 1994; Canadian Association of Petroleum Producers 1993). The effectiveness of these guidelines has seldom, if ever, been evaluated. One reason for this may be the lack of reliable methods and the high cost of testing the guidelines and monitoring compliance. Reliable and consistent methods to appraise guidelines and to perform environmental audits are clearly needed, particularly given current Alberta Government policies to downsize and transfer management responsibilities to resource industries.

Study Area

The study area is located in the Hinton–Edson foothill region of west-central Alberta. The area is primarily forested with pure and mixed stands of lodgepole pine, white spruce, and aspen. Elevations vary from 1000 m near Edson to 2800 m at the Jasper National Park boundary. Climate is characterized as continental, with cold winters and cool summers. Annual precipitation varies from 500 to 550 mm, with approximately 50 to 60% occurring as rainfall in the summer months (Environment Canada 1996). Runoff regimen is dominated by snowmelt, with the greatest proportion of annual flow occurring in the months of May and early June (see Fig. 1).

Soils in the region developed from glacial material and are characterized by lacustrine and aeolian deposits and till material. Soils in general are highly susceptible to erosion. Sediment transport and deposition in streams from road stream crossings and other similar disturbances are common (Rothwell 1983).

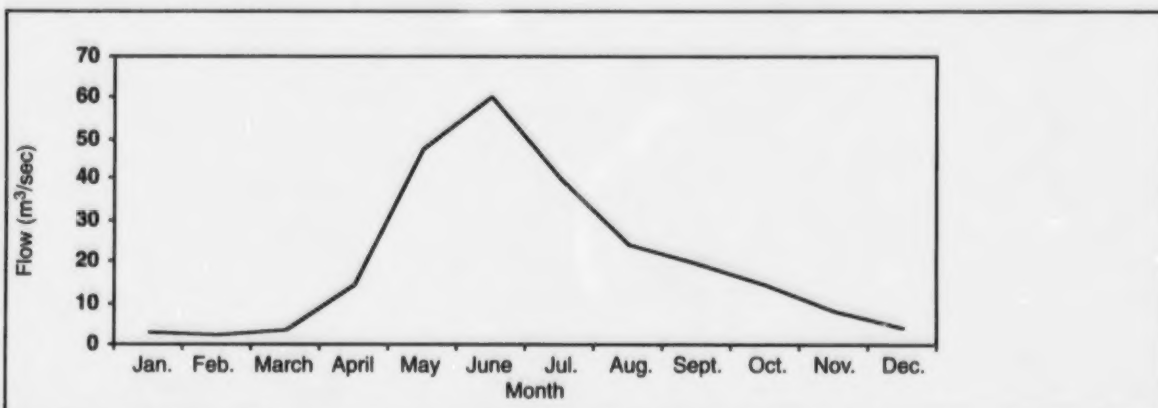

Figure 1. Average annual hydrograph (1954–1973) for the McLeod River above the Embarras River lat. 53°28′10″N, long 116°37′45″ W (Water Survey of Canada 1974).

Major rivers in the region are the Athabasca, McLeod, Berland, and Pembina. These rivers and their tributaries support wild populations of rainbow trout [*Oncorhynchus mykiss* (Walbaum)], bull trout, Arctic grayling [*Thymallus arcticus (Pallas)*], and mountain whitefish [*Prosopium williamsoni* (Girard)] (Nelson and Paetz 1992). The area was selected for study because it has an extensive industrial system of roads and road stream crossings developed over the last 40 to 50 years to support forestry, petroleum, and mining industries.

Objectives

The objective of this paper is to discuss the logic and suitability of freeze-core techniques to sample for sediment intrusion in Alberta foothill streams. We feel that given the early progress of our research, discussion of our methods and the rationale for sampling is a valuable contribution to these proceedings. Full reporting of baseline information on sediment intrusion and information on successful adaptation of freeze-core techniques to local conditions will follow at a later date.

The primary hypothesis selected for testing in this study was "is the degree of sediment intrusion higher downstream of road stream crossings than upstream?" The hypothesis reflects a commonly held belief that sediment intrusion associated with roads has significantly contributed to the deterioration of aquatic habitats (MacDonald et al. 1991; Sterling 1992).

Materials and Methods

Selection of Study Stream Crossings

Several road stream crossings (bridges and culverts) were examined for study and sampling. Criteria used for the selection of road stream crossings were based on surface substrate size and similarity of upstream and downstream reaches. Out of more than 100 crossings observed, only twelve satisfied these criteria. The initial focus of the study was to concentrate on evaluating sediment intrusion in substrate suitable as, or similar to, spawning material for salmonid species endemic to the region. Consultation with local biologists and a review of the literature indicated gravel size 1–4 cm in diameter was a preferred spawning substrate size for local rainbow and bull trout.

Sampling Within the Stream

One to four paired upstream and downstream sample sites were selected at each road-stream crossing based on the presence of substrate suitable for local spawning fish (1–4 cm), and similar streamflow velocity. Selection of sites with similar velocity was assumed to represent similar conditions for transport and deposition of similar-sized sediment and bedload.

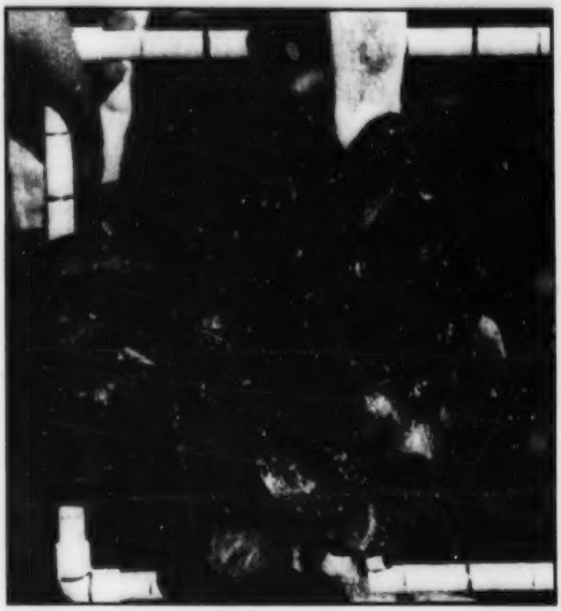

Figure 2. Photograph of extracted freeze-core sample with platy rocks.

Streambed material was sampled using the freeze core method (Walkotten 1976; Everest et al. 1980). The use of a constant volume method, such as the one described by Rood and Church (1994) was not possible because of the presence of large flat rocks horizontally aligned throughout the substrate. These large rocks also made removal of frozen core samples very difficult (see Fig. 2).

The freeze-core samples were obtained by driving a hollow-steel probe, with a case-hardened conical tip into the streambed to a depth of 30 cm. Dry ice was inserted into the probe causing the stream substrate near the probe to freeze and adhere to the probe. Samples were cooled for 25–30 minutes. Once frozen, the substrate sample was extracted by forcibly rocking the probe back and forth until the frozen sample separated from the surrounding unfrozen substrate. Once separated, the sample was lifted out of the streambed. Following removal from streams, the frozen substrate was carefully removed from the probes by use of a cold chisel. Several well-placed strikes with the chisel and hammer were

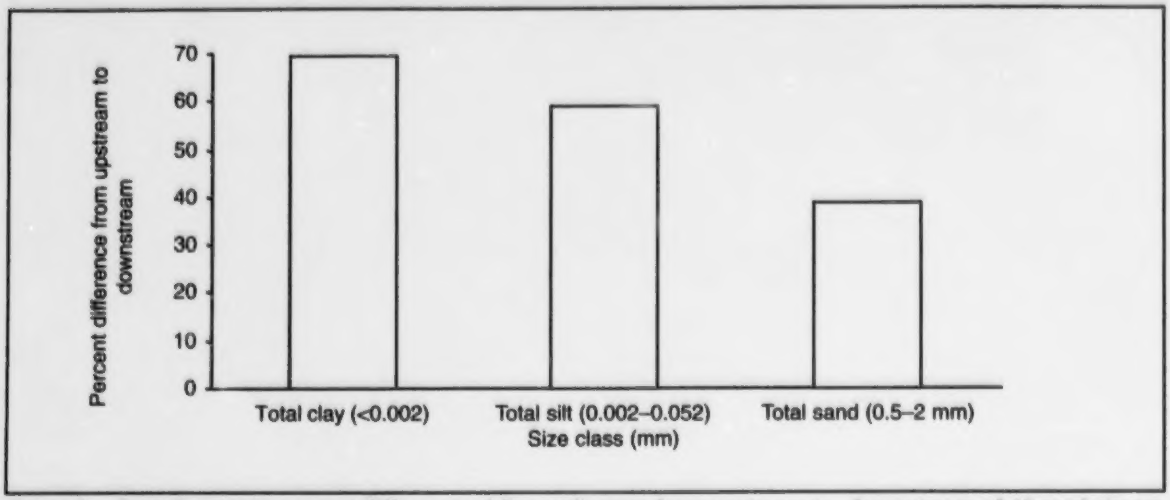

Figure 3. Overall average percent difference of fine sediments from upstream to downstream of 12 road stream crossings.

usually sufficient to fracture the frozen substrate into large pieces that could be bagged and stored while they thawed. Very minimal damage occurred to individual grains or cobbles and very little of the samples were lost by chiseling. The use of a blowtorch to melt the samples was tested but proved ineffective. Samples were then thawed, stored, and later analyzed in the laboratory for fine sediment content.

Sediment intrusion was assumed to occur if the downstream samples contained a higher percentage of fine sediment (less than 2 mm) than upstream samples. For the purpose of this report, all upstream samples for each stream were combined, as were the downstream samples. The comparison between upstream and downstream was calculated as follows

$$\%\Delta S = \frac{(D_A - U_A)}{U_A} \times 100$$

where D_A = percent by weight of sand, silt, or clay for all downstream samples of stream A, U_A = percent by weight of sand, silt, or clay for all upstream samples of stream A, $\%\Delta S$ = percent difference of fine sediment from upstream to downstream samples of stream A.

If $\%\Delta S$ is greater than 0, then more fine sediment is present in the samples taken downstream of the crossing than upstream. In this case it was assumed that sediment intrusion occurred with the crossing being the primary source. If $\%\Delta S$ is less than or equal to 0, then there is less sediment present in the downstream samples than upstream of the crossing. It was assumed, in this case, that sediment intrusion did not occur.

At each sample location, measures of channel width and depth, and distance to the stream crossing were obtained. Scaled black and white photos of *in-situ* substrates were also obtained for each sample location.

Results

Frozen samples were usually 30–40 cm in length, 15–35 cm in diameter, and cylindrical in shape. Samples weighed 11.3 kg on average, with a maximum of 28 kg. Twenty-five percent of the samples were greater than 15 kg. The average weight of combined samples in a stream was 52.8 kg.

Seven out of twelve streams showed a greater amount of fine sediment in substrates (sand, silt, and clay) downstream than upstream. Five streams showed more clay upstream of the crossing than downstream. The overall average for all 12 streams showed more fine sediment in substrates downstream of crossings (Fig. 3).

Stream substrates were highly heterogeneous and consisted of gravels, finer sediments, and often large, platy rocks. Table 1 shows the distribution of average particle sizes for upstream and downstream samples.

There was a greater proportion of particles <25 mm in size downstream of crossings than upstream. In contrast, there was a greater proportion of particles >50 mm upstream of crossings.

Table 1. Average particle size distribution for upstream and downstream samples

	Downstream	Upstream
%>50 mm	14	28
%25–50 mm	22	22
%2–25 mm	44	35
%<2 mm	20	15
Total	100	100

Discussion

The sampling method was effective in detecting differences in sediment intrusion between combined upstream and downstream samples. There were, however, difficulties in selecting enough streams from which to sample. Further, the high variability of substrate within streams will require modification for future sampling, and more intense sampling.

The selection of streams with similar upstream and downstream reaches at road-stream crossings was difficult to achieve. Many stream crossings were characterized by changes in gradients that made upstream and downstream reaches very different and often could not be used. For instance, a channel could be soft bottomed downstream of the crossing because of a low gradient, while upstream the channel was gravel bottomed and steep. Changing gradients were often a reflection of road location on benches or breaks in slope, or might have been caused by the crossing itself. As such, the sample size was limited, and the use of sampling criteria, such as the age of crossing, was not possible.

Preliminary tests displayed differences among streams for sediment intrusion. Only about half the streams show sediment intrusion from the road-stream crossing. Despite the variability, there was overall, more fine sediment downstream. In particular, clay was found in much greater amounts downstream of crossings than upstream. In contrast, it is difficult to explain why there was more clay in the upstream substrate of five study streams. Several factors may contribute to this variation. First, there is extreme variability of substrate material within each stream. Second, outside factors such as the age of the crossing, the degree of reclamation, stability of the stream, and natural sources were not included in this preliminary analysis.

A greater proportion of particles downstream in the 2–25 mm class than upstream may show the addition of gravel from the roads. Higher proportions of gravel and cobble material upstream (>50 mm) may be indicative of the streams' natural state without the addition of particles less than 25 mm. In other words, there may be greater disturbance downstream in terms of fine sediment and material 2–25 mm. Greater deposition due to reduced velocities may also have contributed to more fine material downstream. This can occur from deposition at the crossing outlet creating lower flows. Or, it can occur from plunge pools created from hanging culverts.

Table 2. Summary of sample weights extracted using various methods and coolants (Rood and Church 1994)

Method of sample extraction	Coolant	Weight of extracted samples (kg)
Excavated core methods	Not applicable	6 to 15
Freeze-core sampling with a single tube	Liquid carbon dioxide	1.5–2
Tri-tube corer	Liquid carbon dioxide	up to 20
Single probe	Liquid nitrogen	10–15
Modified, constant volume, freeze-core apparatus	Liquid nitrogen	maximum 13.5

Large distances between the upstream and downstream sample sites were often necessary to satisfy the streamflow velocity criterion for pairing samples. This may have contributed to the variability of samples within streams. Further variability may have occurred because the distance between one pair of samples in a stream may have been large, while another pair in the same stream was close together. To reduce this variability in the future, individual samples will be taken from similar habitat types (i.e., pool, riffle, run) upstream and downstream, and samples will be taken on a transect.

The weight of samples using our methodology (average 11.3 kg) was larger than, or comparable to, methods used by other researchers. Lisle and Eads (1991) used a tri-tube sampler and produced samples often 10–15 kg. A study comparing several substrate extraction methods by Grost and Hubert (1991) yielded samples averaging 1.4 kg by freeze-coring with carbon dioxide gas, 4.8 kg by excavated coring, and 3 kg by shovel extraction. A review by Rood and Church (1994) indicated variable sample weights for different sample extraction methods (Table 2).

A greater weight per sample, or at least a larger number of samples to provide a greater sample weight in a given stream, is desirable to properly represent the full range of substrate sizes and larger cobble sizes in particular (Rood and Church 1994).

Future sample collection for the study will address this concern by leaving the samples to freeze for a longer period of time to increase the average weight per sample, and by extracting more samples per stream.

The trend of sediment intrusion from these road-stream crossings seems evident, but variability and a small sample size are a problem statistically. In order to overcome the problems associated with variability, future sampling and analysis will attempt to increase the sample size, the number of samples, and their size within each stream. A different sampling methodology will be used to reduce variability among sample extraction sites. Individual samples will be taken from similar habitat types (i.e., pool, riffle, run) upstream and downstream. Future analysis will attempt to assess the significance that crossing age, crossing type, and stream and crossing conditions have on the level of sediment intrusion.

Acknowledgments

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Sedimentation Effects on Benthic Macroinvertebrates and Rainbow Trout in a Southern Appalachian Stream[1]

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Abstract

Benthic macroinvertebrate densities and rainbow trout (Oncorhynchus mykiss) catch-rates in a southern Blue Ridge escarpment stream were related to substrate changes caused by sedimentation from construction of a pumped-storage hydroelectric project. Benthos densities and trout catch-rates declined as fine sediments accumulated in the substrate. Young-of-year (YOY) catch-rates for rainbow trout collapsed prior to the decline of adult trout catch-rates, suggesting that spawning had been adversely affected. Very fine sand (VFS) and composite substrate parameters that reflected increasing fine sediments were correlated with the decline in benthos densities and rainbow trout catch-rates. Substrate quality began to recover as sedimentation rates decreased.

[1]Study funded by Duke Power Company, Charlotte, NC.

Van Lear, D.H., Abernathy, A.R., Barwick, D.H., Dysart, B.C., III, Manuel, K.M., Miller, S.D., Dillard, S.M., and Wood, T.H. 1998. Sedimentation effects on benthic macroinvertebrates and rainbow trout in a southern Appalachian stream. Pages 451–453 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

In this expanded abstract, we report the effects of heavy sedimentation on aquatic macroinvertebrates and wild rainbow trout in Howard Creek, a southern Blue Ridge escarpment stream. Although excessive fine sediments are known to have adverse effects on aquatic communities, the issue of what constitutes fine sediments and how benthos and fish respond to substrate changes over extended periods of time are not well documented.

Study Area

Stream biota were monitored prior to and during construction of the Bad Creek Reservoir over an 11-yr period. Howard Creek is a third-order headwater stream in northwestern South Carolina which flows from about 975 m above mean sea level to 335 m elevation at its confluence with Lake Jocassee.

Methods

Composite substrate samples were collected in riffles along transects at four stations from January 1980 through December 1990. A control station (H/9) was located upstream of any project-related activity while three impacted stations were downstream of the project. Sands and smaller particles in the substrate were analyzed by sand fraction analysis (Dysart et al. 1981). Median particle size and fine-to-coarse ratios were used to characterize the entire substrate sample smaller than very coarse gravel.

Benthic macroinvertebrate sampling was conducted 3–4 times/yr at each station using Surber samplers. The relative abundance of young-of-year (YOY) and adult rainbow trout at each station was determined by once-through electrofishing in the fall from 1980 through 1990.

Results and Discussion

Fine sand fractions were the first substrate parameters to respond to sedimentation. Very fine sand (VFS) averaged more than an order of magnitude higher than its preconstruction value during the peak period of clearing and construction activities (1985–1986). Substrate conditions gradually improved from 1987 through 1990 as construction activities slowed and erosion control measures became more effective.

Benthic macroinvertebrate densities declined markedly during construction of the Bad Creek project and mirrored trends in VFS% (Fig. 1). In 1985 and 1986, benthos densities were only 5 and 2%,

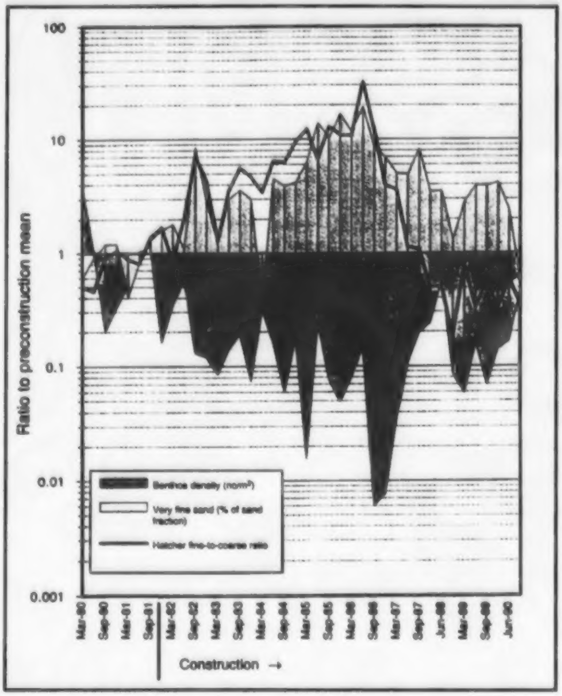

Figure 1. Changes in benthic macroinvertebrate densities vs. changes in the very fine sand percent and Hatcher's fine-to-coarse ratio at station H/7 on Howard Creek.

respectively, of their preconstruction mean. Heavy sedimentation can transform stream bottoms to a habitat unsuitable for many species characteristic of stony-bottomed riffles (Reiser et al. 1989; Wesche et al. 1992). Densities increased toward the end of the study.

Both YOY and adult rainbow trout were adversely affected by heavy sedimentation (Fig. 2). In 1985, no YOY were captured and in 1986, neither adult nor YOY trout were collected. Young-of-year and adult trout catch-rates through 1986 were strongly and negatively correlated ($r = -0.96$ and -0.88, respectively) with yearly averages of VFS%. Fine sediments prevent oxygen from reaching eggs, trap fry in the substrate, and retard removal of toxic compounds from redds (Bjornn et al. 1977; Young et al. 1991; Waters 1995). Without adequate recruitment and because life spans are short in Howard Creek (few trout reach the 3-yr age class), adult rainbow trout populations quickly declined at impacted stations.

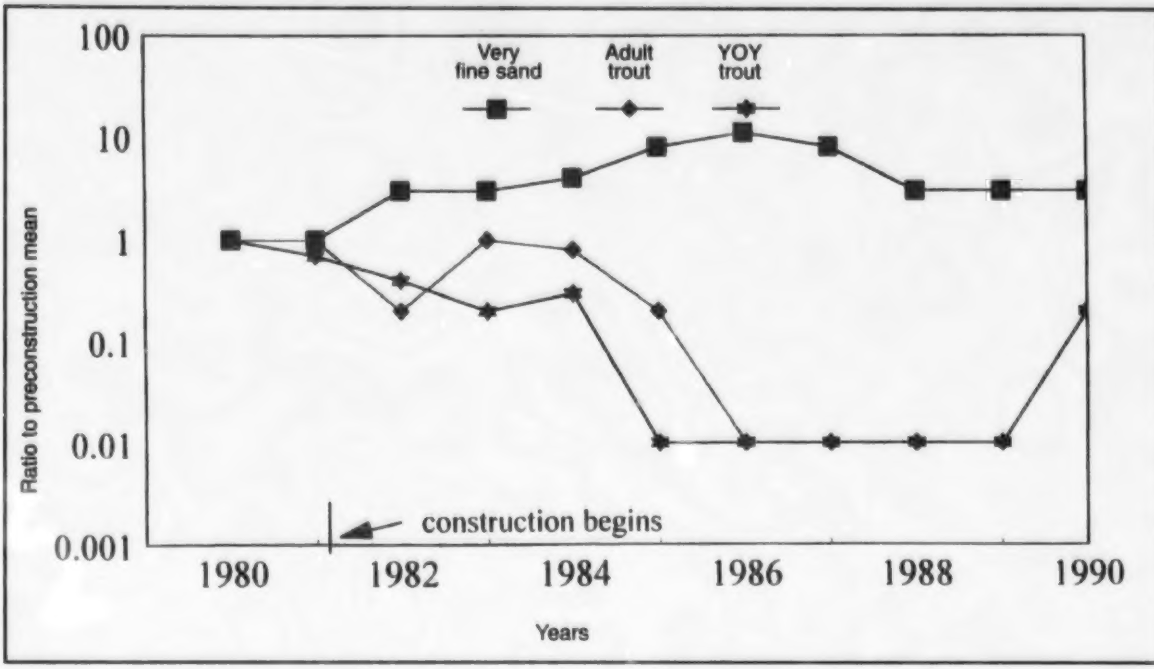

Figure 2. Relationship between very fine sand percent (station H/7) and catch-rates of young-of-year (YOY) and adult rainbow trout (station H/6) in Howard Creek (based on ratio to pre-construction means).

Conclusions

The composition of the substrate of Howard Creek changed through time in response to sedimentation caused by construction activities. The benthic macroinvertebrate community declined dramatically due to increasing dominance of fines in the substrate. Substrate parameters that reflected increasing fines were also related to declining trout catch-rates. Substrate quality improved as construction activities slowed and erosion control measures became more effective.

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Channel Morphology of Small Central Interior Streams: Preliminary Results from the Stuart-Takla Fish/Forestry Interaction Program

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Abstract

The channel morphology of three stream reaches is being studied in each of three forested watersheds in the Sub-Boreal Spruce biogeoclimatic zone of central British Columbia. The objective of these studies is to quantify the impact of forest management activities consistent with the B.C. Forest Practices Code on channel morphology. The selected watersheds have been compared by multivariate analysis and are shown to be suitably matched with respect to biophysical characteristics; this ensures that between-watershed comparisons are valid. The natural (i.e., not influenced by forest management) stream channel morphology of these streams has been poorly documented in the past. The stream reaches vary from low gradient channels with fine textured banks and gravel beds and riffle-pool morphologies to steeper channels with cobble and boulder banks and bed and cascade-pool morphologies. The main factors influencing channel morphology are sediment supply/transport and large woody debris (LWD) characteristics. Both are considered, but the latter is stressed because LWD exerts an important influence in these streams and, in many ways, the functional role of debris is similar to that documented previously in coastal streams. Results are compared to those from channel studies in coastal environments.

Hogan, D.L., Cheong, A., and Hilger, J. 1998. Channel Morphology of Small Central Interior Streams: Preliminary Results from the Stuart-Takla Fish/Forestry Interaction Program. Pages 455–470 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

Studies of stream channel morphology began in 1991 as part of the Stuart-Takla Fish/Forestry Interaction Program. The studies were designed to explore the nature and extent of morphological change in streams resulting from forest management in interior watersheds. There have been numerous studies of forestry-stream channel interactions in coastal watersheds (e.g., Salo and Cundy 1987; Toews and Moore 1982; Hogan 1986). As a result of such papers, new forest practices have been legislated in the province of British Columbia. Unfortunately, relatively little stream channel work has been conducted in the interior of B.C. and less is known about the influence of the different forestry practices on the stream environment. The Stuart-Takla studies will provide a basis for comparison relating the affects in stream channels in interior versus coastal environments and will consider the importance of different forest management practices as well as geographic differences.

The central objectives of the channel morphology studies are to:

- characterize the physical channel conditions typical of central interior streams;
- quantify the impact of forest management activities consistent with the British Columbia Forest Practices Code on channel morphology, and;
- compare and contrast the forestry/channel morphology interactions in coastal and interior environments.

In this paper, in addition to introducing the physical stream studies at Stuart-Takla, we provide preliminary data characterizing the channel morphology of small forested watersheds in the central interior of British Columbia. We will also show the types of analysis that will be completed once logging has occurred.

Background and the Coastal Experience

Numerous coastal fish/forestry studies have documented changes in channel morphology as a result of logging. Many of these have concentrated on altered large woody debris (LWD) characteristics, due to logging, as a primary or partial cause for channel changes. Selected findings from the Queen Charlotte Islands (Fig. 1) are summarized here in order to provide a rational for the design and implementation of the interior channel studies. Except where noted, the following summary is from Hogan (1986).

Typical coastal small stream channels are mapped in Figures 2 and 3. The former is an example of a stream flowing through an old growth coastal western hemlock (CWH) forested watershed (drainage basin area of 16 km^2). This channel is very diverse, with complex longitudinal and planemetric forms. The longitudinal profile, with an average gradient of 1.3%, has distinct, well defined, pools and riffles. Pools, primarily formed by lateral scour, account for almost 65% of the overall channel area. The channel width is highly variable, alternating between narrow and wide sections. Banks are commonly undercut and channel bars consist of cobble, gravel and sand size materials. Large woody debris is prevalent and frequently spans the channel from bank to bank. The predominant orientation of the debris pieces is either perpendicular or diagonal to the general alignment of the banks. Most of the debris has root wads attached to the log trunk.

By comparison, the channel shown in Figure 3 is located in a logged CWH watershed (57% logged during the 1960s by high lead methods without

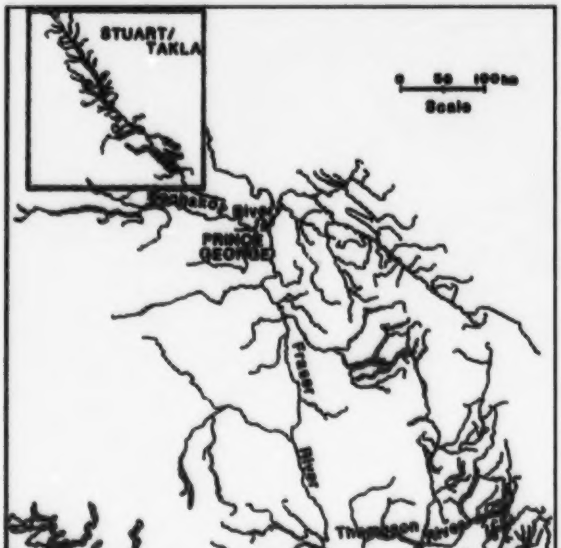

Figure 1. Location map of British Columbia showing the Stuart-Takla Fish/Forestry Interaction Program watersheds and the Queen Charlotte Islands.

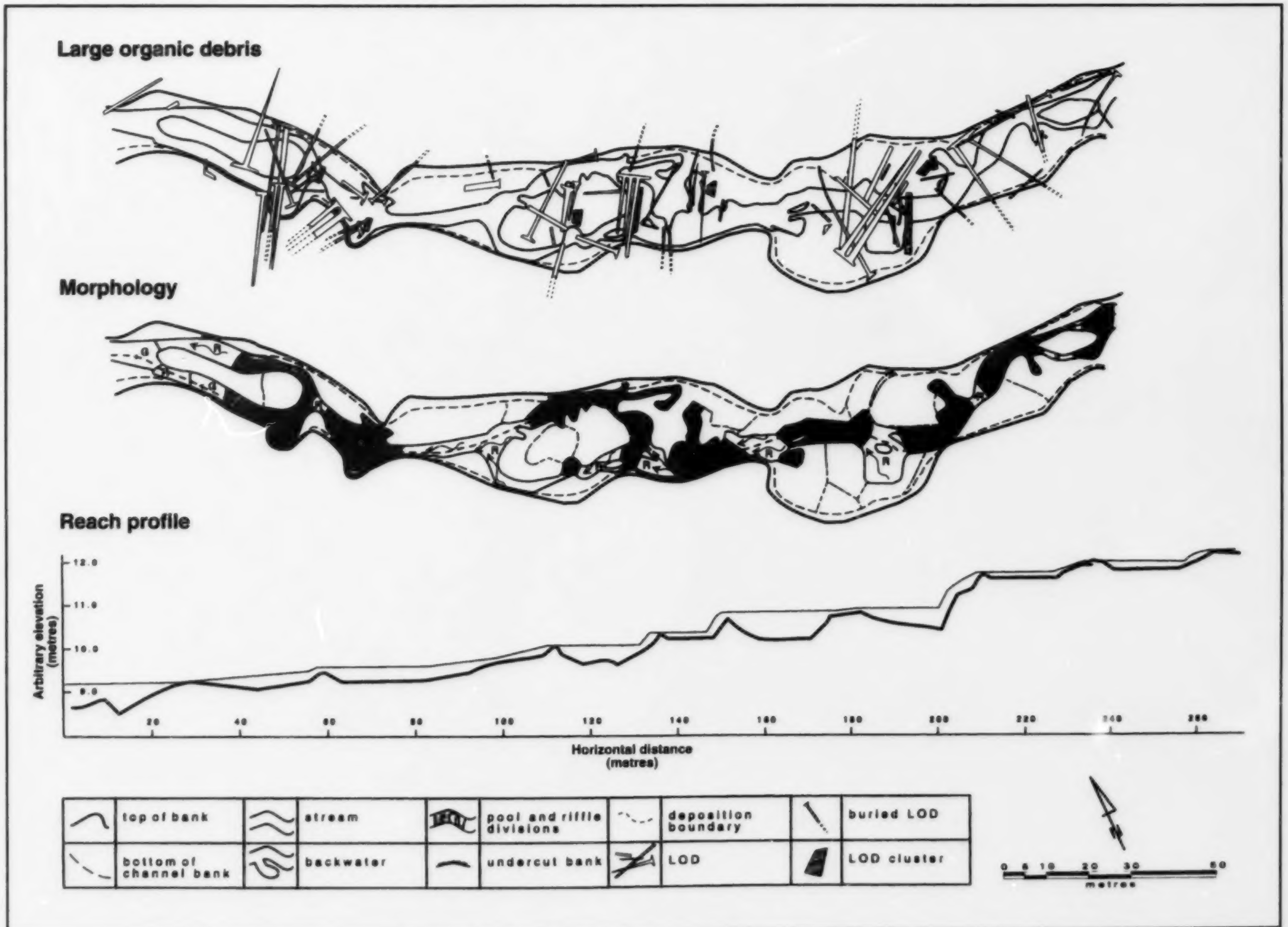

Figure 2. Channel morphology, LWD characteristics and longitudinal profile of an old growth CWH forested watershed stream (from Hogan 1986).

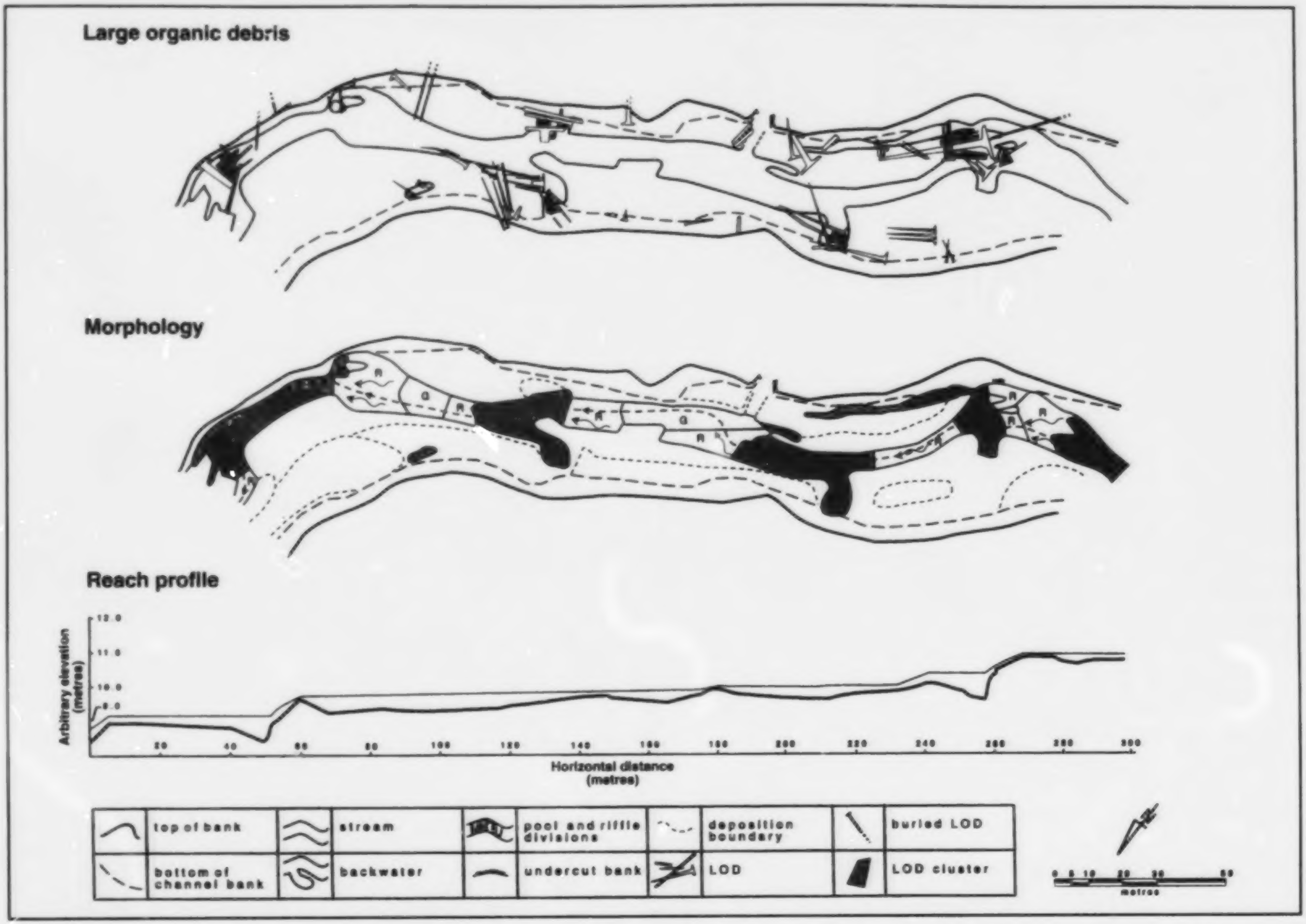

Figure 3. Channel morphology, LWD characteristics and longitudinal profile of an old growth CWH logged watershed stream (from Hogan 1986).

leave strips). This watershed is similar in most morphometric attributes (see "Study design and watershed characteristics" later in this paper) to the unlogged watershed stream mapped in Figure 2. The logged stream channel is relatively simple with minimal variability in longitudinal, planimetric and sedimentologic characteristics. The longitudinal profile shows long pools with relatively uniform depths. Although riffles and glides are slightly more prevalent than in the forested channel (45% riffle and glide in the logged compared to 35%), the shape of each morphological feature is different than in the forested stream. Both pools and riffles are long, narrow and shallow. Channel width is not only wider than expected for the drainage basin size and hydrology, it is consistently wide with minimal variability; undercut banks are absent. Sediment texture variability is also low.

Large woody debris characteristics are significantly different in the logged stream compared to the forested watershed stream. There are lower volumes of debris in the logged channel and the size distribution is shifted towards more abundant small pieces. The most important change is the shift in LWD orientation; there is significantly more debris oriented parallel to the channel banks, compared to the diagonal arrangement in the forested stream. This shift leads to less interaction with the stream flow and sediment transport so the same amount of debris has far less influence on scouring and trapping sediment in the logged stream. There is almost twice as much sediment stored along the logged channel bed and this material is located in fewer than one quarter as many storage sites.

Summarizing the coastal experience, Hogan (1987) found that logging has led to a reduction in bank stability (diminished area of undercut banks), pool area (smaller and shallower due to infilling), LWD steps and flow widths (due largely to aggradation and de-watering). In these logged channels there has also been an increase in riffle (extending into the downstream pool) and channel bankfull width.

The channel studies currently underway at Stuart-Takla are designed to test if channel conditions respond to logging in a similar fashion as those in coastal environments. These studies will also attempt to identify the importance of recently introduced logging prescriptions, which are, in many cases, very different than experienced previously on the coast (Table 1). The studies will concentrate on

Table 1. Basic differences between past coastal and future interior logging practices important to fish-forestry interactions research

1960–80s coast	1990s interior	
Study site	**Study site**	
Steep landslide prone terrain, stream channels closely coupled to hillslopes.	1) Steep landslide prone terrain, stream channels closely coupled to hillslopes.	2) Stable terrain or stream channels not coupled to hillslopes.
Rain dominated hydrology.	Snow-melt dominated hydrology.	Snow-melt dominated hydrology.
Logging (treatment)	**Logging (treatment)**	
Road building practices that greatly increased the frequency and intensity if landslides and sediment delivery to streams.	Professionally engineered roads.	
Logging to stream edge, removal of LWD recruits.	Buffer strips left as recommended by code.	
Possible removal of LWD from stream channels.	No removal of LWD from stream channels.	
Streambank damage from yarding or skidding across stream channels.	No streambank damage in fish bearing streams.	
Large areas harvested leading to possible hydrological impacts.	Application of present practices to limit peak flow increases.	

changes in the variability of specific channel attributes over time; these will include channel width, depth, sediment texture, morphology and large woody debris.

Study Design

The Stuart-Takla Fisheries/Forestry Interactions Project is based on a paired watershed approach (MacDonald 1998 a,b). Three watersheds are being compared. In the future, two will be logged, at different times and rates, to various levels (spatial extent), and by specific harvesting techniques. Although the logging plans have not been finalized, all practices will conform fully with the British Columbia Forest Practises Code. The third watershed will remain unlogged and will provide a control for the two treatment watersheds. All three watersheds are currently forested so management and channel morphology interactions can not be considered at this time. The channels discussed here represent natural, anthropogenically undisturbed streams typical of the central interior of British Columbia.

An implicit assumption of paired watershed studies is that all watersheds to be compared are biophysically similar. This is necessary because a measured difference in a given channel attribute can be assumed to be due to the treatment only if all other factors that influence the attribute are alike. Therefore, to make valid conclusions regarding forestry management and channel morphology, the watershed processes that control channel behavior must be comparable. An attempt is made here to provide a rigorous test of the study watershed's biophysical similarity.

Watershed Characteristics

The general shape and appearance of a stream channel is determined by many factors. Church (1992) suggests the most important of these include:

- flood regime characteristics of the stream;
- amount, timing and nature of sediment and debris delivered to the stream;
- nature of the materials through which the stream flows; and
- local geological history of the area.

Several secondary factors govern channel morphology (Church 1992), including:

- local climate
- nature of riparian vegetation
- human modification of the channel (direct effects), and
- land use (indirect effects).

Most of these factors depend on properties of the watershed, and so must be considered when selecting comparable study sites. Hydrology is considered elsewhere (see Beaudry 1998 and Heinonen 1998). The other factors are considered here.

The testing and grouping procedure developed by Cheong (1996) is used to determine the degree of watershed dissimilarity. In this procedure, a series of 14 topologic and morphometric measures are obtained from standard large-scale NTS maps and aerial photographs. The basin characteristics include those that have significant geomorphic importance and have been shown to have a direct influence on sediment and woody debris delivery to the stream channel (Table 2). Two key factors, percent valley flat (<7%) and percent steepland (>60%), relate to sediment delivery and to the nature of material through which the stream flows.

The degree of similarity (called the dissimilarity index) between watersheds is determined by grouping the watershed attributes and comparing the Euclidean distance separating them. The dissimilarity index values for the study sites range from 1.9 to 10.6 (Table 3). Identical watersheds score a zero and larger values indicate greater between-watershed differences. Review of several hundred dissimilarity indices, gathered in all areas of British Columbia, shows that only 2% of the values fall below 9 (Cheong 1993). This value is the current provincial standard for determining similar watersheds (Cheong 1996). The testing conducted here confirms that the study watersheds are physically similar.

Other watershed characteristics are also considered. The forest types are the same in all three watersheds (Table 2). The riparian vegetation along each stream is also alike (Table 4), as are the geological materials (Ryder 1995).

Channel Characteristics

Stream channels are to be compared within and between watersheds over the duration of the study. Therefore, after selecting comparable watersheds, it is necessary to ensure that the streams to be compared have similar physical channel characteristics. It is essential that only channels with similar morphologies and planimetric form are compared because channels respond differently to land-use changes. For instance, a step-pool morphology channel (typical of headwater streams) will not generally

Table 2. Watershed characteristics

	Watershed				
	Gluskie	Forfar	O'Ne-ell main	O'Ne-ell sub	Tsitsutl
Biogeoclimactic zone					
upper basin	SBS[a]	SBS	SBS	SBS	SBS
lower basin	ESSF[b]	ESSF	ESSF	ESSF	ESSF
Drainage basin area (Ad) (km^2)	51.2	38.4	76.4	40.3	31.1
Main channel gradient	0.038	0.064	0.066	0.075	0.069
Lake area (% of Ad)	0.2	0.1	0.4	0.3	0.1
Valley flat (% of Ad)	38.0	7.0	4.2	1.1	0.3
Steepland area (% of Ad)	8.2	0.1	0.8	0.8	0.8
Mean basin gradient (m/m)	0.191	0.185	0.202	0.19	0.23
Drainage density (km/km^2)	0.78	0.64	1.08	0.92	1.25
Relief (m)	1250	1320	1265	1255	1255
Mean elevation (m)	1310	1310	1210	1245	1270
Drainage magnitude	13	13	26	15	9

SBS[a] = sub-boreal spruce.

ESSF[b] = Engleman spruce - subalpine fir.

Table 3. Testing the biophysical similarity of watersheds. Results are Dissimilarity Index values obtained from multivariate analyses (Cheong 1996), based on 14 morphometric variables (see partial set listed in Table 4).

	Watershed				
	Gluskie	Forfar	O'Ne-ell main	O'Ne-ell sub	Tsitsutl
Gluskie	—[a]	—	—	—	—
Forfar	6.26	—	—	—	—
O'Ne-ell main	10.59	6.28	—	—	—
O'Ne-ell sub	8.10	2.07	4.78	—	—
Tsitsutl	6.44	1.94	7.86	3.30	—

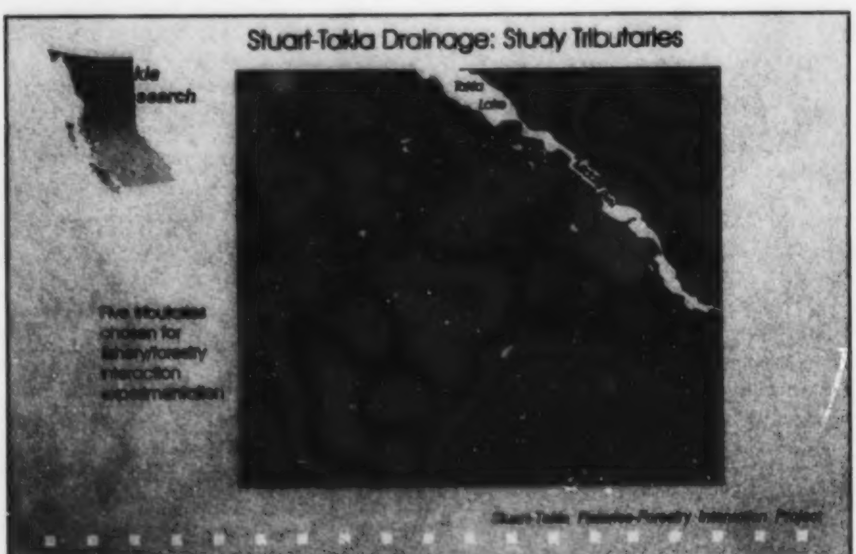

Figure 4. Map showing location of the stream morphology study reaches.

Table 4. Distribution of riparian plant species

a) Overstory

Species	Forfar			Gluskie			O'Ne-ell		
	F1	F2	F3	G1	G2	G3	O1	O2	O3
Hybrid white spruce (*Picea glauca* X *Engelmannii*)	F	F	f(D/S L SIDE)	F	F	F	F	F	F
Subalpine fir (*Abies lasiocarpa*)	vf	f	vf	f	f	S	f	f	S
Lodgepole pine (*Pinus contorta*)			F						
Black cottonwood (*Populus balsamifera* Ssp. *Trichocarpa*)	S	vf				f			
Trembling aspen (*Populus tremuloides*)			f	f					
Paper birch (*Betula papyrifera*)				f	S	vf			
Sitka mountain-ash (*Sorbus sitchensis*)		S	f			f		vf	S
Sitka alder (*Alnus viridis*)	F	F	F	F	F	F	F	F	F
Willows (*Salix Sp.*)	F	F	F	F	F	F	F	F	F

b) Understory

Species	Forfar			Gluskie			O'Ne-ell		
	F1	F2	F3	G1	G2	G3	O1	O2	O3
Vaccinium Sp. Red-osier dogwood (*Cornus stolonifera*)	F	F	F	F	F	F	F	F	F
Black twinberry (*Lonicera involucrata*)	S	F	S	S	F	S	F	F	S
Black gooseberry (*Ribes lacustre*)	S	S			S	S		f	S
Skunk currant (*Ribes glandulosum*)		f		S	f	vf			
Highbush-cranberry (*Viburnum edule*)	S	f	S	S	S	S		F	S
Thimbleberry (*Rubus parviflorus*)	f	S	S		F	S		F	S
Devil's club (*Oplopanax horridus*)	S	F	F	F	S	F	f	F	F
Red raspberry (*Rubus idaeus*)	S	F	f	f	F	S	F	f	
Prickly rose (*Rosa acicularis*)	f	S			S	F		f	f

Species	Forfar			Gluskie			O'Ne-ell		
	F1	F2	F3	G1	G2	G3	O1	O2	O3
Red elderberry (*Sambucus racemosa*)	S	F			F	S	S	S	F
Saskatoon (*Amelanchior alnifolia*)									f
Fireweed (*Epilobium angustifolium*)	S	S	f	f	S	f		S	
Cow-parsnip (*Heracleum lanatum*)	F	S		S	F	S	F	S	S
Stinging nettle (*Urtica dioica*)	S	S			S	f	F	S	f
Fern Spp.:									
Lady fern	S	F	f		F	f	F	F	S
Oak fern	S	S	f		S	vf		S	f
Horsetail Spp.:									
Wood horsetail (Equisetum sylvaticum)			f						
Common horsetail (E. arvense)	S	S		F	S	f	S	S	S

Legend: F = Frequent; S = Scattered; f = Few; vf = Very few.

Table 5. Morphological characteristics of the study stream reaches

Reach characteristic	Gluskie			Forfar			O'Ne-ell		
	G1	G2	G3	F1	F2	F3	O1	O2	O3
Bankfull width, Wb	14.8	11.16	12.85	14.2	10.64	11.22	14.4	12.28	11.2
Reach length, L (m)	236	184	122	190	192	138	219	156	132
Gradient (%)	0.7	1.3	1.4	0.6	1.4	2.0	0.7	1.6	2.0
Sediment texture	gravel/sand	gravel/cobble	cobble/gravel	gravel/sand	gravel/cobble	cobble/gravel	sand/gravel	gravel/cobble	cobble/gravel
Depth (m)	1.30	1.21	1.54	1.03	1.35	1.06	1.51	1.28	1.5
Pool length (% of L)	78	57	62	62	62	51	85	57	51
Riffle length (% of L)	0	25	15	0	5	28	0	9	24
Run length (% of L)	22	19	23	38	33	21	15	34	25
Pool/Riffle ratio (W_b)	2.3	2.1	1.9	1.7	1.8	3.8	4.4	1.4	1.2
Number of log steps	3	3	-	5	5	-	1	3	-

become wider as a result of a disturbance because the channel banks commonly consist of non-alluvial, erosion resistant materials. Conversely, a riffle-pool morphology channel with alluvial banks will widen as a result of increased sediment loads or direct mechanical disturbance of the bank (Hogan 1987). It is not useful to compare the response of these two types of streams. Similarly, it is not prudent to compare a straight channel with one with bends.

Three stream channel reaches have been selected for study in each of the three watersheds. The location of the nine reaches is shown in Figure 4; the reaches furthest downstream are identified as stream name, reach 1 (e.g., Gluskie Reach 1 is labelled as G1) and the furthest upstream are identified as stream name, reach 3 (e.g., Gluskie Reach 3 is labelled as G3).

The reaches closest to the stream outlets are low gradient channels with fine textured fluvial and

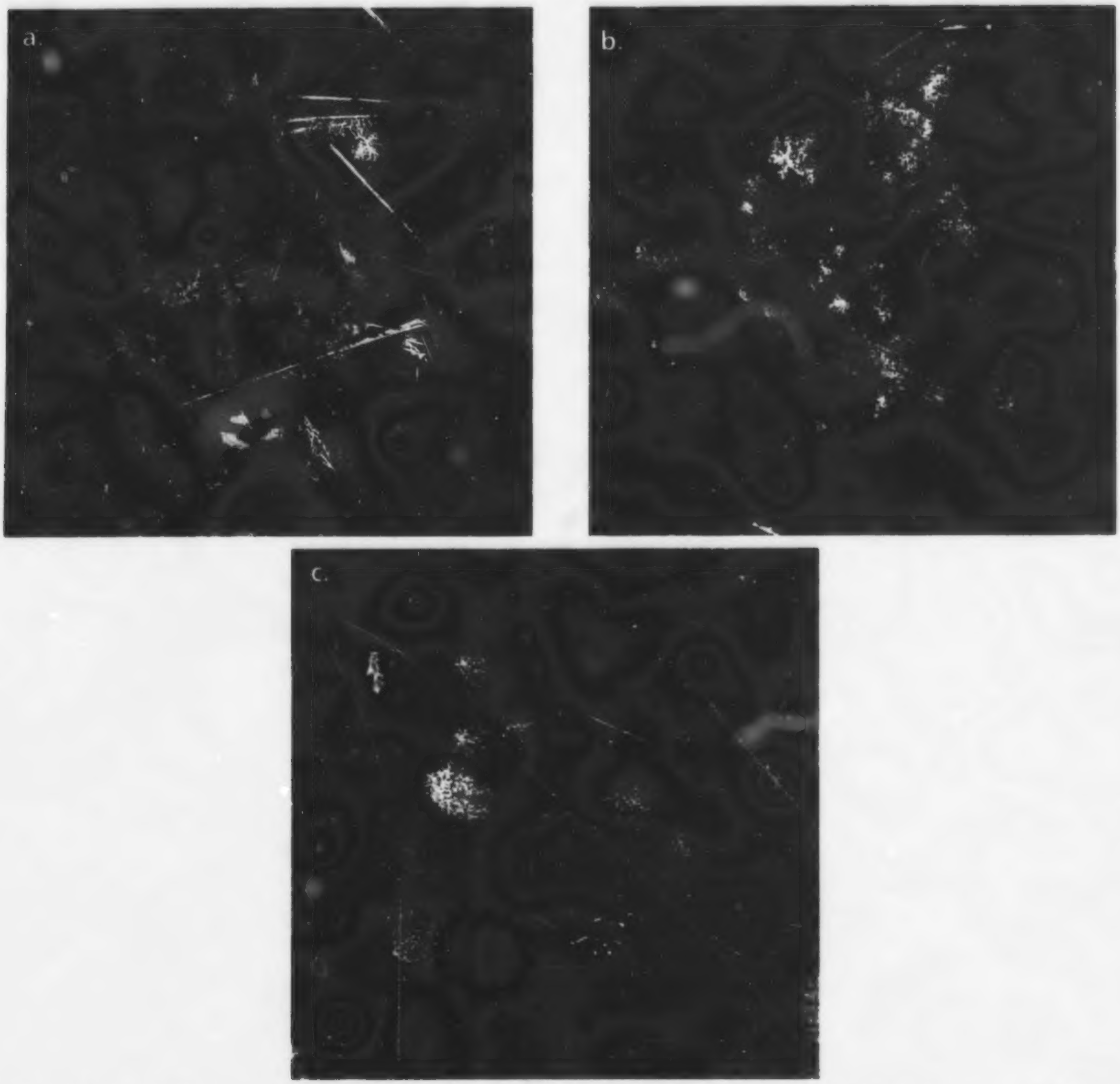

Figure 5. Low level aerial photographs (taken in September 1992 using a large format Hasselblad camera at about 100 m altitude) of the three study reaches in O'Ne-ell Creek. The blue ribbons crossing the stream channel indicate the location of cross-sectional survey transects (see Fig. 9).

lacustrine bank sediments. The middle and upper reaches are progressively steeper with coarser textured bed and bank materials. The physical and morphological characteristics of each reach are given in Table 5. Low level aerial photographs of the three study reaches in O'Ne-ell Creek provide examples of the different stream environments (Fig. 5).

The reaches to be compared after logging has occurred are Gluskie Reach 1 (G1) with Forfar Reach 1 (F1) and O'Ne-ell Reach 1 (O1) in one group, G2, F2 and O2 in the second group and G3, F3 and O3 in the third group. The channels are very well matched in all cases. Channel width, gradient, depth, sediment texture and pool-riffle characteristics are close to identical (Table 5). It is not appropriate to mix the comparisons; that is, it is incorrect to compare G1 with F2, for example.

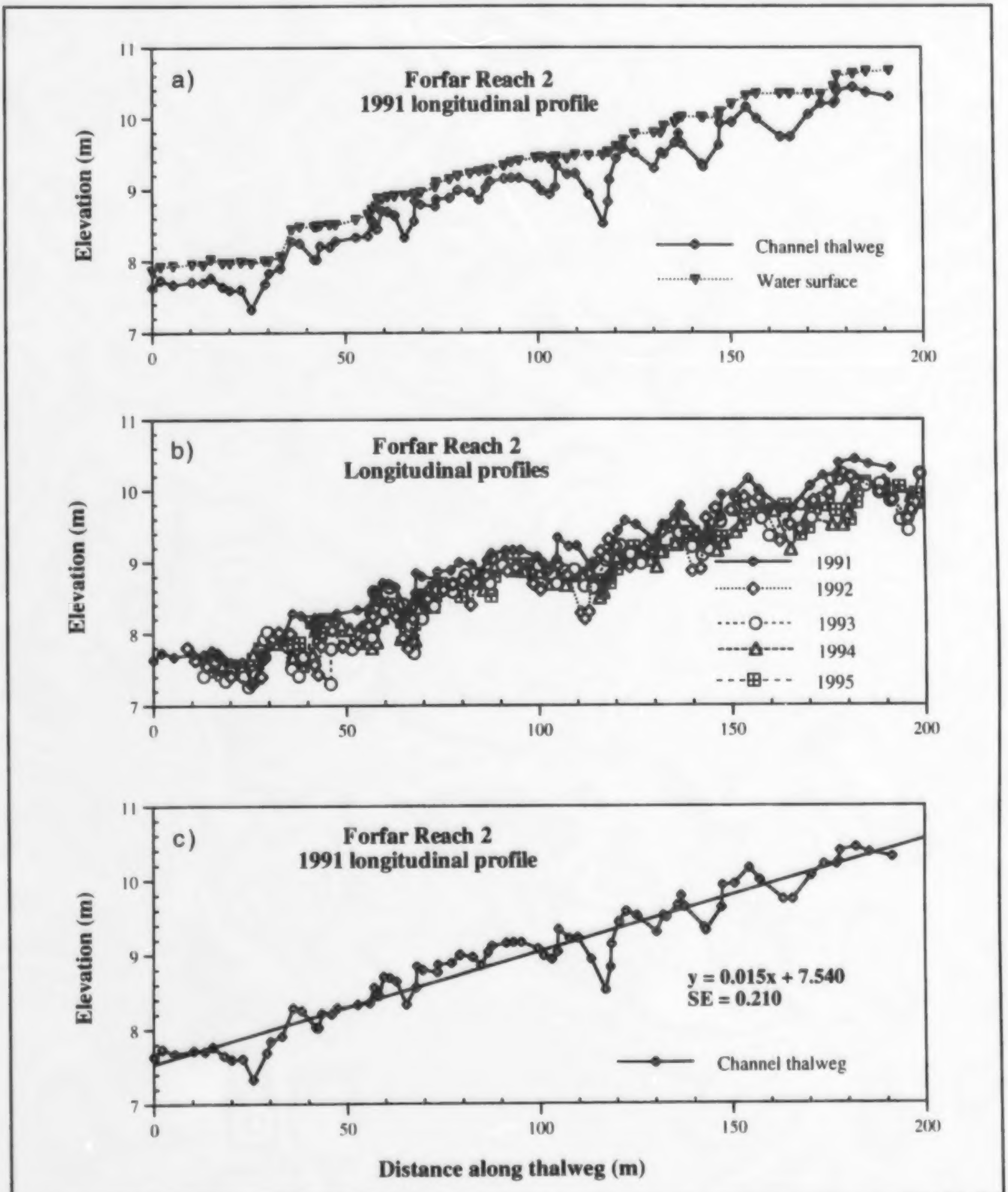

Figure 6. Longitudinal profiles for Forfar Reach 2.

Channel Measurements

Standard field survey techniques are used to measure channel conditions. An automatic level, stadia rod and distance chains are used for the surveys. Each reach has 11 monument cross-sections, separated by one bankfull width (W_b), and topographic, sedimentologic and vegetation features are surveyed each year (usually in September). Longitudinal surveys are completed to document annual changes in the bed profile. Each 1 W_b segment of the channel is also photographed annually from fixed ground sites. Low-level aerial photography is taken using a helicopter-supported, large-format (70 mm) camera platform. The aerial photographs are used in a Carto Analytical Plotter (AP 190) to provide planimetrically correct channel details. Maps (1:500 scale) are drawn from all of these sources for each reach and for each year of the study.

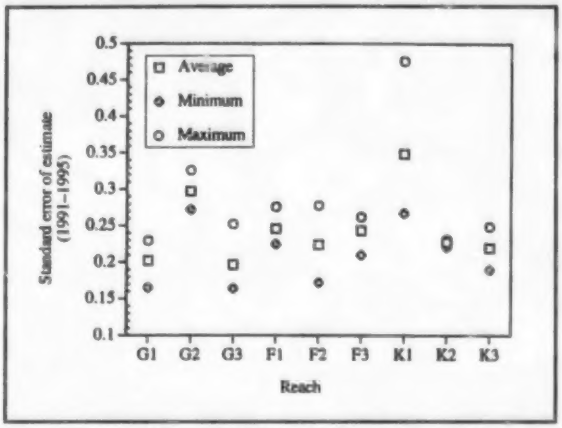

Figure 7. Depth variability of the study reaches on Stuart-Takla Streams.

Channel Morphology

Coastal studies indicated that significant changes occurred in depth, width and pool-riffle shape variability after logging. Preliminary results from the central interior streams provide an indication of the amount of variability present naturally. Only data from selected reaches are presented for the sake of brevity (this is appropriate because logging has not occurred and the channels are so similar).

The longitudinal profile of F2 for 1991 (Fig. 6a) shows the typical riffle (shallow and steep) and pool (deep and flat) morphology. To show changes over

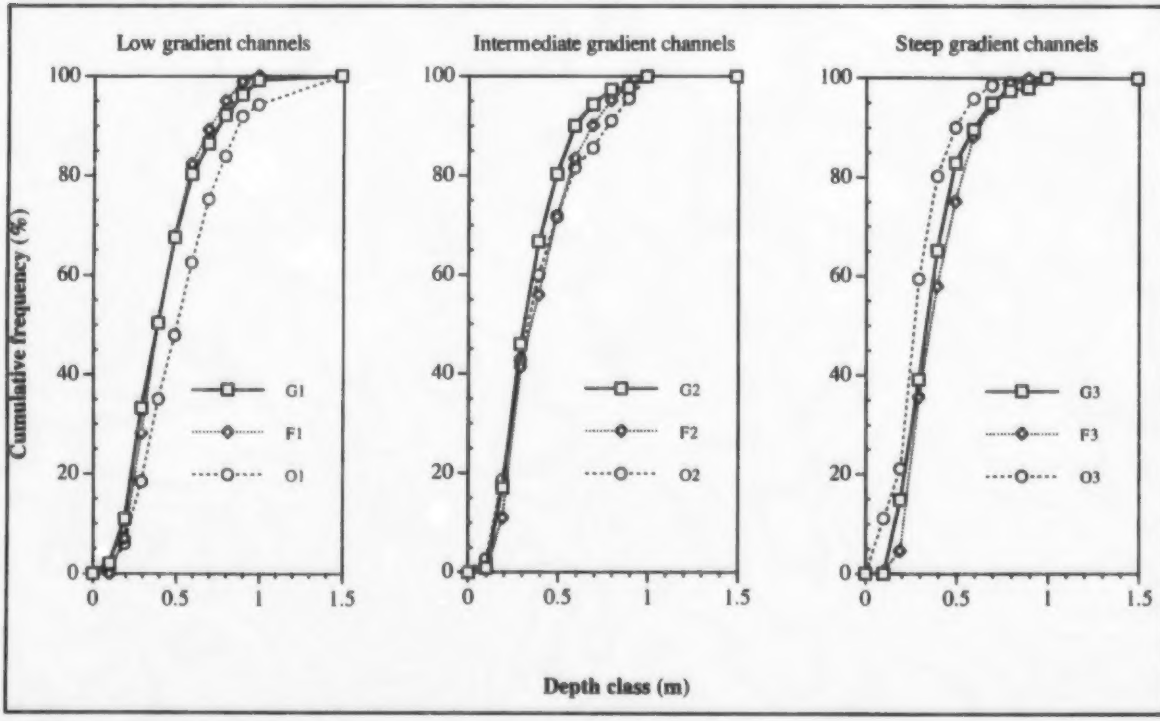

Figure 8. Depth frequency plots.

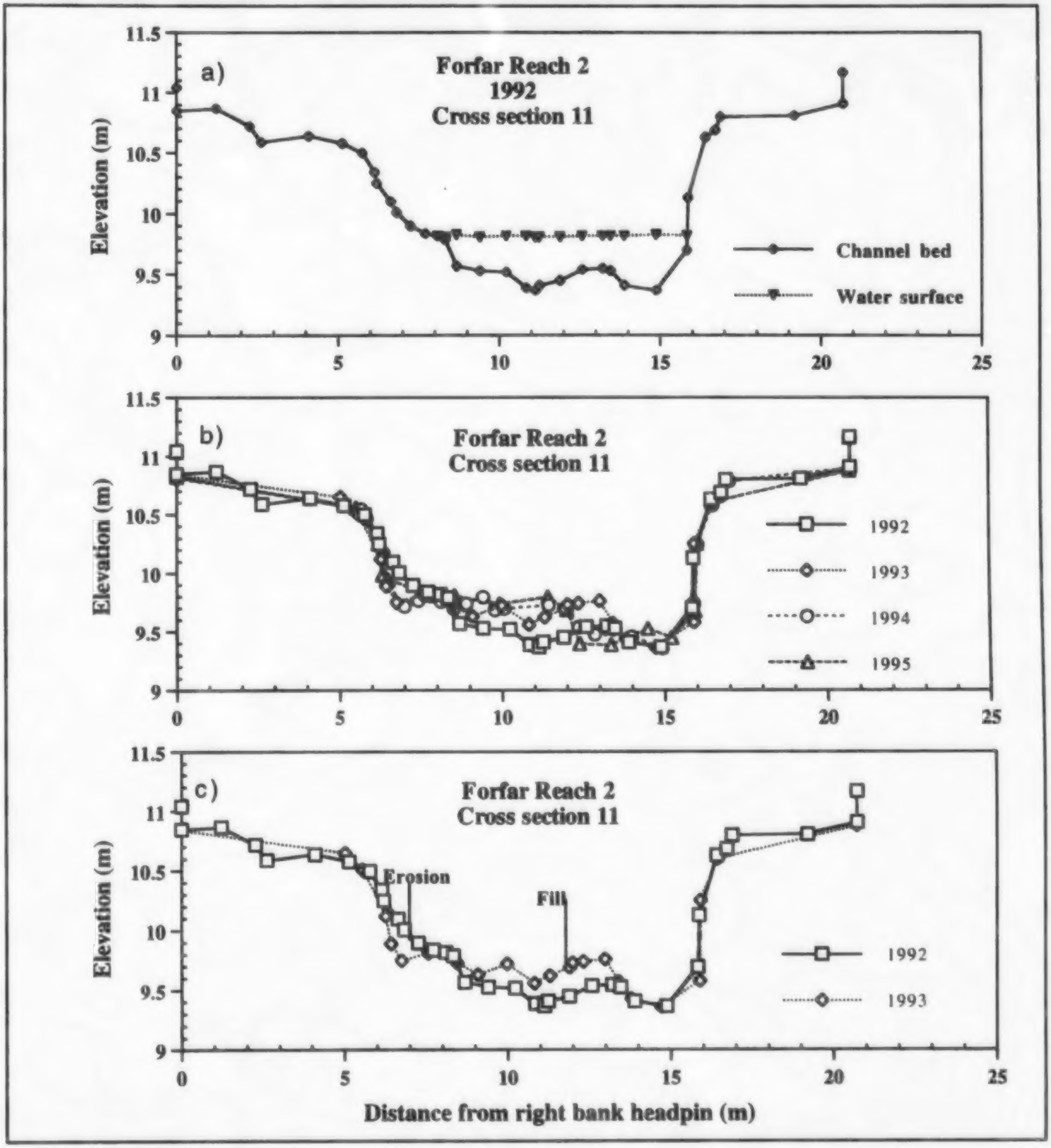

Figure 9. Cross section plots for Forfar Reach 2.

time it is common to plot all of the years as overlay plots (Fig. 6b) but these are often difficult to interpret. The variability of the longitudinal profile can be quantified by calculating the standard error of estimate (SE) for a regression line drawn through the thalweg profile (the SE is the standard deviation of the residuals around the calculated regression line). An example of the longitudinal profile and SE value is given in Figure 6c. If the profile becomes less variable over time, or as a consequence of logging as suggested by the coastal study results, the SE value will be reduced. The average and minimum/maximum ranges for all SE values are shown in Figure 7. There is a high degree of variability in channel depths in these natural streams; any changes occurring during and after logging will be detected.

The proportion of channel with specific flow depths during low streamflow conditions is often of interest to fish habitat biologists. The depth frequency plots (Fig. 8) show a consistent pattern. In the steeper channels (G3, F3 and O3), about 50% of the channels have depths less than or equal to 0.2 m and less than 10% have depths exceeding 0.6 m. The larger, lower gradient channels (G1, F1 and O1) have 50% of the channel shallower than 0.4–0.5 m. In these reaches, more than 20% of the channel lengths are deeper than 0.6 m and depths do exceed 1.0 m, although infrequently (<5% of the channel).

The monument cross sections in each reach are used to evaluate channel width characteristics. A typical cross section, using F2, is given in Figure 9 (note that the cross sections are visible (blue flagging tape) in Figure 5). Changes in width over time are assessed by plotting multiple years of the same cross section on single plot (Figure 9b). In this example,

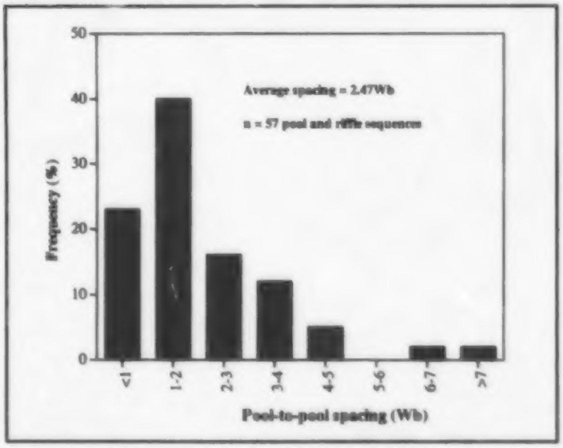

Figure 10. Pool-riffle spacing distribution.

and in all others, there has been no change in width over the five-year period (1991–1995). This indicates that the banks are stable and that erosion is infrequent and limited in spatial extent.

The cross-section data are also used to measure sediment accumulation, or fill (bed aggradation), and erosion, or down-cutting (bed degradation) over time. The annual cycle of aggradation and degradation evident in cross section 11 in F2, shown in Figure 9c, is normal for these channels. The total volumes of sediment eroded and deposited along the reaches have not yet been calculated. This will be done by using all of the cross sections, in addition to the large scale maps and photogrammetrically rectified low-level aerial photographs, once logging has occurred in the watersheds.

The pool-riffle spacing frequency distribution is skewed toward short lengths (Fig. 10). The average spacing distance, for all streams combined, is 2.5 W_b, although the pool-riffle ratios for individual reaches range from 1.2 to 4.4 (Table 5). These values are similar to those reported for coastal forested watershed streams (Hogan 1987).

Large woody debris inventories are included in Table 6. The total number of pieces in each reach is relatively high, ranging from 0.0319 to 0.0822 pieces per unit channel area. By contrast, the range in four forested coastal streams in the CWH zone is 0.024–0.048 pieces/m^2 (Hogan 1986). However, the size distribution is also similar to the coast with more small and intermediate sizes. With only one exception (O2), all reaches have at least 10% of the material in the large size class.

The origin of in-stream LWD is often of interest because new regulations aim to protect the source of LWD (e.g., British Columbia Forest Practices Code regulations). This is difficult to accomplish if the source is not known. Leaving buffer strips along a channel to provide future sources of LWD works only if debris comes from the streambank. Blowdown and bank collapse delivery of debris to the study streams is important, averaging about 20% of the total debris load in all reaches (Table 6). In most cases, however, the majority of the LWD is derived from upstream sources. This indicates that riparian zones in upstream zones, beyond those areas used by anadromous fish, should be protected if the long term source of debris is to be maintained.

The LWD in the study streams is mostly stable and is not transported frequently. Over 70% of all debris in each stream is classed as either stable or very stable (Table 6). Further, most of the LWD

Table 6. Large woody debris characteristics

Large woody debris	Gluskie G1	Gluskie G2	Gluskie G3	Forfar F1	Forfar F2	Forfar F3	O'Ne-ell O1	O'Ne-ell O2	O'Ne-ell O3
Total number of LWD pieces (N) in reach	138	134	50	173	168	100	153	128	113
Pieces per unit area (N/m^2)	0.0676	0.0653	0.0319	0.0641	0.0822	0.0646	0.0485	0.0668	0.0764
Size (% of N)									
Small 0.016–0.63 m^3	59	49	65	45	35	60	54	63	55
Moderate 0.63 –2.8 m^3	19	33	25	17	29	16	36	30	30
Large >2.8 m^3	23	18	10	36	36	25	10	7	15
Origin (% of N)									
Blow-down	28	18	6	41	40	5	18	9	23
Bank collapse	9	24	13	13	13	28	11	6	10
Floated in from U/S	63	58	81	46	47	67	71	85	67
Stability (% of N)									
Unstable	37	25	30	16	21	25	22	31	11
Stable	16	20	50	9	28	21	24	29	41
Very stable	47	55	20	75	51	54	54	40	49
Decomposition (% of N)									
Very rotten	52	49	59	56	20	32	13	21	39
Old	46	36	38	37	56	68	82	75	55
Fresh	2	15	3	7	24	0	5	4	6
Pieces missing from prev. year (mean % of N)	9	11	4	14	12	6	7	8	2

(>80%) is badly decomposed and less than 10% is moved out of the reach each year. Overall, the LWD inventory is consistent with the other morphological indications that the channels are stable.

Conclusion

The stream channel morphology component of the Stuart-Takla Fish/Forestry Interaction Program is designed to document the riverine environments of British Columbia's central interior watersheds. The study includes nine reaches selected to represent a range of channel sizes and morphologies. The study channels are satisfactorily matched, with respect to biophysical attributes, to allow valid between watershed comparisons after logging has been completed.

The natural, unlogged watershed streams are characterized by very diverse channel morphologies. These include highly variable depths, widths, sediment textures and woody debris loadings. Although there are spatial differences along the streams, the channels are very stable, with only minor annual variations in channel bed, bank and LWD characteristics. The conditions indicate that a long time period has elapsed since the last major channel-disturbing event occurred. If this natural stability persists until logging is undertaken in the treatment watersheds then any influence of forest management practices will be more definitively identified. If, on the other hand, a large natural disturbance occurs before logging, it will be more difficult to isolate the influence of forest management.

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Effects of Forest Management on Westslope Cutthroat Trout Distribution and Abundance in the Coeur d'Alene River System, Idaho, U.S.A.

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Abstract

A stratified random electrofishing survey was employed to sample 42 second order and 31 third order streams within the Coeur d' Alene River basin, Idaho, U.S.A., during 1994 and 1995 to estimate densities of westslope cutthroat trout (*Oncorhynchus clarki lewisi*). Nine 30-m sections in each stream were sampled using single-pass electrofishing, and one of these locations was randomly selected for three-pass electrofishing. We estimated capture efficiency (mean = 68.8%, range = 25% to 100%) for each three-pass location using a maximum likelihood estimator, and through linear regression determined that wetted stream width significantly affected efficiency. We used the estimated sampling efficiency model (E = -0.053*WW + 0.855) to estimate cutthroat trout density and 95% prediction intervals (PI) for single pass locations. Densities ranged from 0.003 fish/m^2 to 0.606 fish/m^2 (mean 95% PI of ±0.197 fish/m^2). Estimated mean density of westslope cutthroat trout was significantly lower for tributaries of the North Fork of the Coeur d' Alene River (0.038 fish/m^2) than that for tributaries of the Coeur d' Alene River (0.152 fish/m^2). Positive correlation was found between pool frequency and cutthroat trout abundance. Pool frequency was negatively correlated to watershed road density. Lowest densities of cutthroat trout populations coincided with areas of the most intense forest management and the lowest pool frequency. We believe that bedload movement into pool habitat during channel forming winter flows may be one factor contributing to reduced cutthroat trout abundance in the Coeur d' Alene River basin. However, all statistical relationships between selected measures of forest management activities were weak and accounted for low amounts of variation in cutthroat trout abundance.

Dunnigan, J. L., Bennett, D. H., and Rieman, B. E. 1998. Effects of Forest Management on Westslope Cutthroat Trout Distribution and Abundance in the coeur d'Alene River System, Idaho, USA. Pages 471-476 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

Four varieties of cutthroat trout (*Oncorhynchus clarki*) are found in Idaho, U.S.A., waters (Simpson and Wallace 1978). One of those, the westslope cutthroat trout (*O. c. lewisi*), is native to the Salmon River drainage and major systems north of the Salmon River, Idaho. Westslope cutthroat trout are highly sensitive to exploitation, habitat alteration, and hybridization with other species (Rieman and Apperson 1989). In Idaho, where the cutthroat trout is the state's official fish, most varieties have been recently managed with restrictive regulations and many populations have responded with increased abundance and size composition. One population in the Coeur d' Alene River has not responded favorably to restrictive regulations, suggesting potential habitat related problems.

Historically, westslope cutthroat trout were abundant in the Coeur d' Alene River basin (Maclay 1940). The combined effects of overfishing, logging activities, road construction, and mineral extraction have reduced numbers of cutthroat trout within the drainage (Lewynsky 1986; Hunt and Bjornn 1992). Bowler (1974) concluded that over-harvest by anglers had reduced abundance of westslope cutthroat trout to low levels, and that populations within the Coeur d' Alene River were not responding to special regulations as well as those in similar systems, such as the St. Joe River and Kelly Creek, Idaho (Bowler 1974).

Westslope cutthroat trout populations in the Coeur d' Alene River system exhibit three life history forms: adfluvial, fluvial, and resident (Bowler 1974; Lewynsky 1986; Hunt and Bjornn 1992). Adfluvial stocks of cutthroat trout historically migrated upriver from Coeur d' Alene Lake and existed throughout most of the river basin. Currently, the adfluvial stock is restricted to the lower portion of the Coeur d' Alene River. Cutthroat trout in tributaries and headwaters of the Coeur d' Alene River are believed to exhibit the resident-type life cycle (Hunt and Bjornn 1992). Spawning of all cutthroat trout stocks is believed to occur within headwater streams (Lewynsky 1986).

Most research indicates that the fluvial stock of cutthroat trout in the Coeur d' Alene River is relatively depressed (Bowler 1974; Lewynsky 1986; Hunt and Bjornn 1992). Hunt and Bjornn (1992) estimated 107 fish/km in the Coeur d' Alene River and 19 fish/km in the North Fork of the Coeur d' Alene River. Although these studies have examined the abundance of cutthroat trout within the North Fork of the Coeur d' Alene and Coeur d' Alene rivers, little is known about cutthroat trout abundance within third order and smaller tributaries within the basin, and how their abundance is affected by physical habitat characteristics within each watershed.

The objectives of our study were to estimate absolute abundances of westslope cutthroat trout in most third order and second order tributaries within the Coeur d' Alene River basin; assess effects of physical habitat characteristics on cutthroat trout abundance; and assess effects of forest management on cutthroat trout abundance. Our goal of sampling a large proportion of the drainage area required us to develop a method for single pass electrofishing to determine absolute densities instead of using methods that require labor-intensive, multiple-pass removals from each sampling location (Zippin 1958; Lobon-Cervia and Utrilla 1993).

Study Area

The Coeur d' Alene River originates on the Pend Oreille divide near the Idaho-Montana state borders, flows southwesterly for approximately 190 km, and drains into Coeur d' Alene Lake. The Coeur d' Alene River basin, encompassing the North Fork of the Coeur d' Alene, and the Coeur d' Alene rivers, has an area of approximately 2,280 km^2 (Fig. 1). Quality of stream habitat within the Coeur d' Alene River basin has been affected by road construction, timber harvest, and mineral extraction since the beginning of the century (Maclay 1940). Maximum road densities in some portions of the lower Coeur d' Alene River basin exceed 19 km/km^2 (US Forest Service, Coeur d' Alene, Idaho, unpublished data). Additionally, the physiographic and geomorphic processes within the Coeur d' Alene River basin contribute to quantity and quality of cutthroat trout habitat within the region. Weathered belt-type geology is dominant and influences the hydrologic processes within the basin (Kappesser 1993).

Discharge of the Coeur d' Alene River has been recorded directly upstream from the confluence of the South Fork and the Coeur d' Alene rivers since 1939 and averaged 53.9 m^3/s (US Geological Survey 1990). Flows usually peak in April and May. However, channel forming flows have occurred from November through February approximately every 2 years in the past 20 years (US Geological Survey 1990). During January of 1974 and 1990, peak flows of 172.7 and 747.6 m^3/s, respectively were recorded.

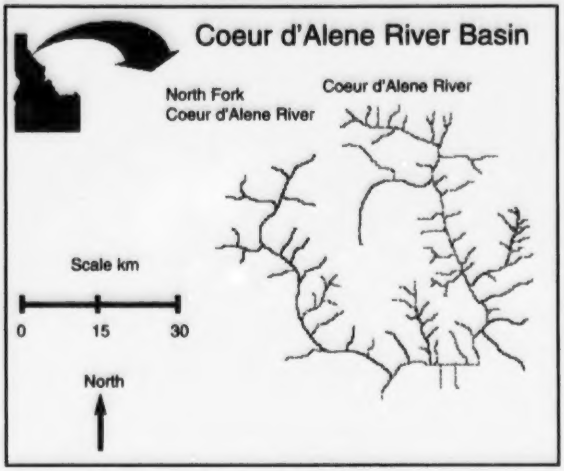

Figure 1. Map of the Coeur d'Alene River basin (area 2,280 km^2), including the North Fork of the Coeur d'Alene and the Coeur d'Alene rivers, Idaho, U.S.A.

Materials and Methods

Sampling

We conducted electrofishing surveys on 42 second-order and 31 third-order streams within the Coeur d' Alene River basin during June through September of 1994 and 1995. Although we tried to sample as many streams as possible, streams were chosen, in part, by accessibility. The length of each stream was measured on a 7.5 minute quadrangle US Geological Survey topological map and divided into three reaches of equal length. Each reach was divided into potential 30-m sampling locations and three locations from each reach were randomly selected for single pass electrofishing. One of the nine locations within each stream was randomly selected for sequential removals with three-pass electrofishing to estimate absolute abundance of cutthroat trout (Zippin 1958; Lobon-Cervia and Utrilla 1993). Sampling was conducted with a Smith Root Model 12 POW backpack electrofisher. Although block nets were not used, we generally ended the sampling location at a change in habitat type.

We measured wetted stream width (minimum of five locations at each location) and bank-full discharge widths at each sampling location. Stream temperatures were recorded (°C) using a pocket thermometer at approximately 08:00, 12:00, and 16:00 hours. The largest range in water temperature was 4°C during the day and from 4°C to 15°C during the sampling season. Stream gradient (nearest 0.5%) was measured at each location with a hand-held clinometer. Visual estimates of habitat complexity, which included cover from large woody debris (LWD, > 30 cm diameter and 1 m long), undercut bank, aquatic vegetation, stream depth, substrate, and turbulence, were made ranging from low complexity (one) to high complexity (ten) at each sampling location. A single investigator determined habitat complexity at all locations to minimize the subjectivity of this estimate. A 500-ml water sample was taken at each sampling location and conductivity was measured (μs.cm^{-2}) using a YSI SCT model conductivity meter. The US Forest Service has been collecting physical habitat data since the early 1980s, including habitat typing inventory (Hankin and Reeves 1988), riffle stability indices (Kappesser 1993), and LWD counts. If physical habitat data existed for streams we sampled, they were included in our analyses. Road densities (km/km^2) in each watershed sampled were calculated using a Geographic Information System. Records of timber harvest activity were obtained from the US Forest Service (US Forest Service, Coeur d' Alene, Idaho, unpublished data) to calculate cumulative clear-cut area for each watershed. To determine the percent cumulative clear-cuts area for each watershed, the area of all timber harvest activity that occurred over the past 30 years in each watershed was summed and divided by the total area of each watershed.

Statistical Analysis

Estimates of population abundance and capture efficiency of cutthroat trout were calculated using a maximum likelihood estimator (Van Deventer and Platts 1983) for locations sampled with multiple passes. Age-0 fish were removed from all samples used to calculate cutthroat trout densities in an attempt to minimize seasonal differences in abundance among streams. Estimates of capture efficiency calculated using the maximum likelihood estimator (Van Deventer and Platts 1983) were compared by multiple linear regression (Ott 1993) to stream gradient, conductivity, wetted stream width, bank-full to wetted stream width ratio, and estimates of habitat complexity to predict capture efficiencies (E). A 95% prediction interval (PI) was calculated and is presented as a measure of variation:

Equation 1.

$$E = \pm\, t_{\alpha/2} \times S_e \sqrt{1 + \frac{1}{n} + \frac{(x - \bar{x})^2}{S_{xx}}}$$

Where $t_{\alpha/2}$ = the t score for 0.05/2 and df = n-2, S_e = standard error of the *E*, and

Equation 2.

$$S_{xx} = \sum x^2 - \frac{(\Sigma x)^2}{n}$$

Mean capture efficiency was 0.69 (± 0.176 SD) and ranged from 0.25 to 1.00 for 71 multiple pass locations sampled during 1994 (0.67 ± 0.161 SD) and 1995 (0.70 ± 0.190 SD). Using the most significant model, the general linear model for predicting sampling efficiency (*E*) for a given wetted stream width (WW) was calculated with:

Equation 3.

$$E = -0.053 * WW + 0.855$$

We then used this model to adjust all single pass catches to estimate densities of westslope cutthroat trout.

Multiple linear regression (Ott 1993) was used to assess the relationship between abundance of cutthroat trout and the physical habitat variables of pool frequency (percent pool habitat compared to all other habitat types), LWD counts, riffle stability indices, cumulative clear-cut area, watershed road density, and stream conductivity. A correlation matrix was calculated to determine how pool frequency, cumulative clear-cut area, and watershed road density were related. Effects of forest management on pool frequency were assessed using multiple regression by correlating cumulative clear-cut area and watershed road density to pool frequency. A multivariate analysis of variance (MANOVA) was used to test for simultaneous differences in abundance of cutthroat trout, pool frequency, watershed road density, and cumulative clear-cut area between streams of the North Fork and Coeur d' Alene rivers (Johnson and Wichern 1992).

Results

Predicted densities of cutthroat trout for all streams sampled during 1994 and 1995 ranged from 0.003 to 0.606 fish/m^2 and averaged 0.125 fish/m^2 (95% PI = ±0.197). Streams of the North Fork of the Coeur d' Alene River basin (n = 20) exhibited a lower mean density (0.038 fish/m^2; ±0.409 SD and mean 95% PI = ±0.034) than streams sampled in the Coeur d' Alene River basin (n = 52; 0.152 fish/m^2; ±0.822 SD and mean 95% PI = ±0.224). Streams in the North Fork of the Coeur d' Alene River had significantly lower abundance than those in the Coeur d' Alene River based on our simultaneous testing (MANOVA) for a difference between abundance of cutthroat trout and pool frequency, watershed road density, and cumulative clear-cut area in each watershed ($p = 0.0217$ using Wilks' Lambda; Johnson and Wichern 1992).

We found no significant relationship ($p > 0.05$) between abundance of cutthroat trout and pool frequency ($p = 0.568$), riffle stability indices ($p = 0.664$), cumulative clear-cut area ($p = 0.491$), road density ($p = 0.833$), LWD counts ($p = 0.769$), stream conductivity ($p = 0.668$), and watershed area ($p = 0.910$). However, we found a significant relationship when the model was reduced to abundance of cutthroat trout and pool frequency ($p = 0.01$). The correlation analysis suggested that pool frequency was negatively correlated to watershed road density ($r = -0.275$; $p = 0.058$) but not cumulative clear-cut area in each watershed ($r = -0.168$; $p = 0.258$). The general linear model for predicting pool frequency from road density was: Pools = - 0.926* Road Density + 33.0 ($r^2 = 0.08$; $p = 0.059$).

Discussion

Estimated mean densities of cutthroat trout in second and third order streams in both the North Fork of the Coeur d'Alene River (0.038 fish·m^{-2}) and Coeur d'Alene rivers (0.152 fish·m^{-2}) were within the range of densities observed by several investigators for westslope cutthroat trout. The higher densities (0.606 fish·m^{-2}) we observed were as high as some reported by others for westslope cutthroat trout. Irving (1987) reported highest densities of ages 1 and older westslope cutthroat trout of 0.3 fish·m^{-2} based on snorkeling in the Priest Lake drainage, Idaho. Thurow (1976) reported highest densities of age 1 and older westslope cutthroat trout from snorkeling in tributaries of the St. Joe River, Idaho, as 0.03 to 0.06 fish·m^{-2}. Lukens (1978) found densities of all age classes of westslope cutthroat trout in Wolf Lodge Creek and its tributaries at about 1 to 2 fish·m^{-2}. Pratt (1984) reported densities of juvenile westslope cutthroat trout ranging from 0.024 to 0.60 fish·m^{-2} in the upper Flathead River basin, Montana. Our lowest densities (0.003 fish·m^{-2}) were slightly lower than Thurow (1976) reported for Marble Creek (0.004 fish·m^{-2}), a tributary of the St. Joe River.

Numerous factors have been related to the abundance of trout in streams. Pool habitat has been identified as critical habitat for cutthroat trout (Schlosser 1991; Irving 1987; Pratt 1984). We found that pool frequency was significantly correlated with cutthroat trout abundance on a stream-level basis. However, Lanka et al. (1987) indicated that geomorphic variables could predict trout standing crops as accurately as physical habitat variables. Kozel and Hubert (1989) suggest that stream size and reach gradient were two dominant geomorphic factors that most strongly influenced stream habitat and cutthroat trout standing crop in streams that were minimally altered by man. Our results showed that these variables were not significant in the Coeur d'Alene River basin, possibly because of the long history of land development activities in the basin (Maclay 1940). Our lowest densities of cutthroat trout generally coincided with conditions on the North Fork of the Coeur d'Alene River of lowest pool frequency, highest cumulative clear-cut area, and highest road density, however, collectively these variables were not significant.

The importance of pool habitat by cutthroat trout is not clearly understood, although its importance on the Coeur d'Alene River system may be related to over-winter conditions. Peak runoff flows occur in April and May, however, bank full flows from rain on snow events can occur during November through February (US Geological Survey 1990) and may be important to overwinter survival. The amount of bedload transported during high flows in the Coeur d'Alene River basin can be as much as 80% to 100% (Rieman and McIntyre 1993). Kappesser (1993) suggests that bedload in the Coeur d'Alene River system tends to deposit in low velocity locations, such as pools. Under conditions such as these, bedload movement into pool habitat may be one factor affecting cutthroat trout abundance in the Coeur d'Alene basin.

Angler harvest has been shown to be another factor influencing the abundance of cutthroat trout in the mainstem Coeur d'Alene River (Bowler 1974; Lewynsky 1986). During this study, we seldom observed fishing in second and third order tributaries due to dense brush cover that occurs adjacent to most tributaries. We believe fishing has little effect on abundance of populations in second and third order tributaries in the Coeur d'Alene River basin.

The influence of forest management activities on trout abundance is not clear (Johnson et al. 1986). Few studies have addressed which component of logging, the canopy removal or the associated road construction, has had the greatest indirect impact on salmonids. Eaglin and Hubert (1993) found that both culvert density and the percent ground surface that had been logged were both positively correlated to cobble embeddedness and fine sediment. Trout standing crop was also negatively related to culvert density (Eaglin and Hubert 1993). Johnson et al. (1986) concluded that any positive effects of canopy removal from clear-cut was offset by a reduction in critical winter-rearing habitat for juvenile steelhead trout (*Oncorhynchus mykiss*). Murphy et al. (1981) concluded that effects of sedimentation could be offset by positive changes in trophic status that increased primary productivity and subsequently increased abundance of salmonids. We believe that the erosive nature of the weathered belt-type geology and subsequent loss of pool habitat was probably not offset by increases in cutthroat trout productivity in tributaries of the Coeur d'Alene basin.

Our results indicate that the lowest cutthroat trout densities generally coincided with the conditions on the North Fork of the Coeur d'Alene River of lowest pool frequency, highest cumulative clear-cut area, and highest road density. However, results from multiple regression analysis relating cutthroat trout density to pool frequency, riffle stability indices, cumulative clear-cut area, road density, LWD counts, stream conductivity, and watershed area revealed no significant relationships. Only after the model was reduced to pool frequency was a significant relationship with cutthroat trout density detected. Pool frequency was weakly correlated with watershed road density but not with cumulative clear-cut area. The weak relationships suggest model saturation, the potential for undetected colinearity among variables, or the potential for spatial autocorrelation; an analysis is yet to be conducted. Also, although we sampled a large number of streams ($n = 73$) and across watersheds larger than 2000 km^2, the wide variation among watersheds may have contributed to our inability to account for greater variation in the densities of westslope cutthroat. Another limitation of sampling as many tributaries as time permits is relying on single pass electrofishing to estimate abundance, although we are confident in the density estimates based on our mean efficiency (0.69), relatively tight 95% prediction intervals, and intensive random sampling of nine 30-m sections in each stream sampled.

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Integrating Timber Harvest with Aquatic Management

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Abstract

Protection of aquatic ecosystems from logging activities commonly involves the use of riparian buffer strips along water courses that are intended to intercept surface-eroded soils, excessive surface water flow, and nutrients from reaching receiving waters (lakes and streams). Vegetation in riparian areas provide benefits including recruitment of large woody debris, shade, litter inputs, and bank stabilization. The use of buffer strips along water courses has arisen as a constraint management tool, principally in the Pacific Northwest. This region is characterized by high levels of annual precipitation, steep and erodible hillslopes, which, in some cases, may cause slow regeneration of conifer species. Unlike coastal regions, Alberta-Pacific's Forest Management Agreement Area (FMA), located in northeastern Alberta, is characterized by low annual precipitation, topographically flat or gently rolling hills, and lakes which receive little surface inflow. The dominant natural disturbance regime in the FMA is fire, which may leave natural buffer strips of various size, or no buffers at all, along water courses. As a result, the frequency distribution of pulses of nutrients and material (e.g., large woody debris) into these lakes and streams is likely important to the ecology of the water course. This paper will discuss the shift that Alberta-Pacific has begun, away from traditional management paradigms that include sustained yield forestry, traditional wildlife management, and riparian buffer strip management. Logging in Alberta-Pacific's FMA is moving toward partial removal, dispersed cutting that incorporates rate and spatial distribution of harvest in watersheds instead of standardized buffer strip management and ecosystem management instead of traditional forest and wildlife management. These new paradigms in forest management are based on a natural disturbance model, where logging activities are designed to approximate the spatial distribution and dynamics of fire. Many of the ideas presented in this paper are preliminary in nature and are currently undergoing rigorous scientific evaluation and validation.

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Aquatic Ecosystem Management under Traditional Forestry Practices

Intensive fiber extraction, which involves clear cutting in 80–90% of the harvesting across Canada, occurs under sustained yield forest management and can result in substantive changes to adjacent aquatic ecosystems. These effects include increased stream flow, sedimentation/siltation, organic debris, changes in nutrient regime, alteration in the thermal properties of streams, and loss of critical spawning habitat for resident or migratory fish populations (Swanson and Hillman 1977; Nicholson et al. 1982; Ahtainen 1992; also see review by Belt et al. 1992). These changes are expected to be more pronounced in regions that receive high annual precipitation, where hillslopes are steep and easily eroded and in situations where forest regeneration is slow (e.g., the Pacific Northwest of North America).

To mitigate or eliminate adverse effects of clear-cut logging on watercourses, jurisdictions utilize riparian buffer strips along watercourses. The regulations governing the width of buffer strips can be either site specific (see Stevens et al. 1995 for a discussion of criteria for British Columbia) or based on a one-size-fits-all concept. In Alberta, for example, buffer strips around small streams and large rivers are 30 and 60 m, respectively, while buffer strips around lakes are 100 m. Under current guidelines, the leave material tied up in riparian buffer strips in the Alberta-Pacific FMA represents about 5 to 10% of the annual allowable cut. The guidelines for buffer strip widths in Alberta are not based on ecological principles or scientific study, but were designed solely to keep heavy logging equipment from operating in stream courses.

A number of factors that led to the creation of buffer strip management in coastal areas of Canada may not apply to northeastern Alberta or in many other parts of interior North America, where fire has been a dominant component in the ecological shaping of ecosystems. While conditions in coastal regions of Canada may necessitate riparian buffer strips, dramatic differences in climate, topography and regeneration rates of trembling aspen (the primary tree species logged by Alberta-Pacific) in northeastern Alberta generate questions about the applicability of traditional aquatic ecosystem protection paradigms. Annual precipitation in the Alberta-Pacific FMA is less than one tenth that of coastal regions: typically <500 mm, with evaporation exceeding precipitation. Additionally, slope in cut block areas is usually 3 to 5% or less. Alberta-Pacific avoids logging on higher slope areas, for both safety and erosion control reasons. Lastly, the regeneration of aspen is much more rapid occurring immediately in the spring after winter logging, at 20–30 000 stems/ha, compared to northern conifer. Therefore one would expect a much more temporary response in water yield changes in streams near aspen-dominated cut blocks (e.g., see Swanson et al. 1998), such as those in Alberta-Pacific harvest areas. The ability to extrapolate and apply a standard buffer width, or a formula, from one province to another, or even one region to another (e.g., from the Alberta foothills to northeastern Alberta) is doubtful. Therefore a new model for lake and stream management is needed in this region of Canada. The key objective of our current research program and management planning is to allow harvesting practices to more closely emulate natural disturbance regimes.

The use of standardized buffer strips also does not account for the time of year in which logging occurs. Soil erosion and sedimentation problems in coastal areas of Canada, associated particularly with summer-time road construction and, to a lesser extent, logging, are likely much reduced or eliminated in northeastern Alberta due to timing of our harvest and road construction activities. For example, 60 to 80% of the roading and logging activities on Alberta-Pacific's FMA occur in winter. This not only improves access to the cut blocks for fellerbunchers, skidders, and logging trucks, but also likely reduces soil compaction and erosion on frozen soils. Differences between summer and winter logging and seasonal road construction need to be investigated.

Buffer strip management in areas of intensive clear cutting may also have a negative effect on riparian forested areas. In certain regions undergoing intensive clear cutting, it is conceivable that current aquatic management regimes may ultimately result in the restriction of old growth forest to riparian areas only. This would greatly reduce the natural structural variability of the forested landscape. The effects of such a spatial distribution of forest on aquatic ecosystems are unknown.

To begin to address some of these concerns, Alberta-Pacific is currently involved in several collaborative studies with Canadian universities and government agencies to examine the utility of using buffer strips for the protection of aquatic ecosystems in northeastern Alberta. Within the TROLS (Terrestrial, Riparian, Organisms, Lakes, and Streams) program based at the University of Alberta, researchers are evaluating the potential protective

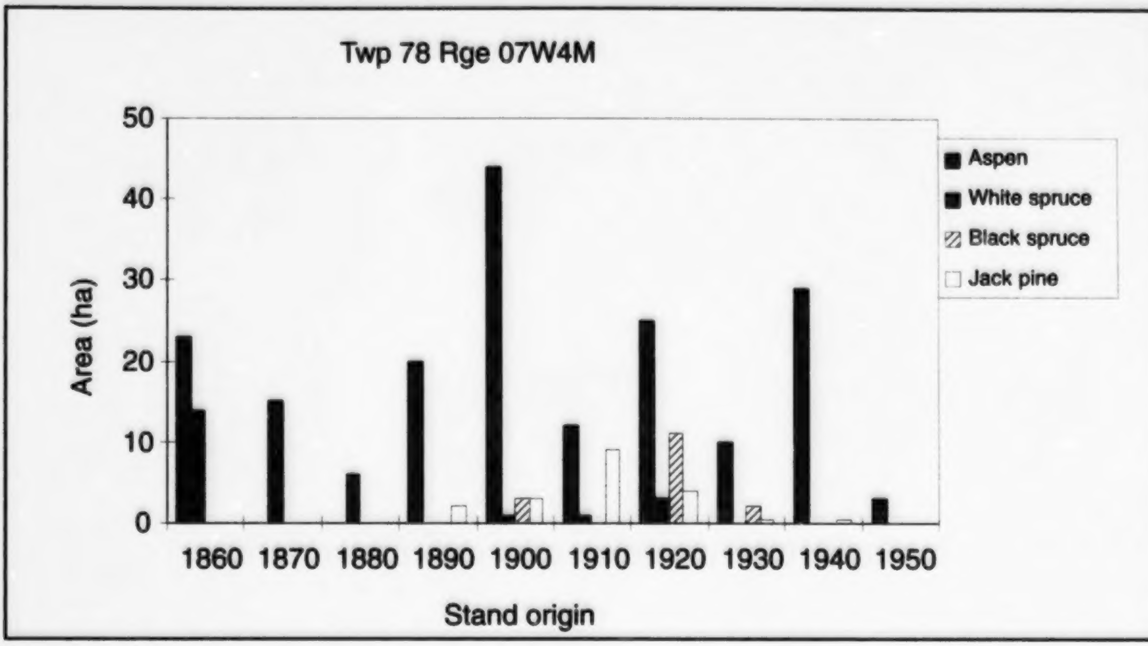

Figure 1. Species composition and age structure of riparian forest within 60 m of the Christina River in northeastern Alberta. Data was compiled from Alberta Forestry AVI information.

effects of differing buffer strip widths around lakes. In each of three regions (Lac La Biche, South Calling Lake, and South Pelican Hills) four lakes will have different buffer strip width treatments applied: 20, 100, 200 and 800 m. The 800-m treatment results in no logging in the lake watershed, thus acting as a reference watershed. In each lake watershed, lakes, as well as the land–water interface between lakes, streams, and associated uplands, will be studied, using variables that include: surface water hydrology and nutrient export, water chemistry in the lakes, phytoplankton, zooplankton, fish communities, and food web linkages. In addition, research will also focus on responses of terrestrial plants and animals (birds, small mammals, etc.) to logging and the varying riparian buffer strip widths.

Also within the TROLS program, we will be evaluating buffer strips along small permanent streams. Five streams within our FMA have been chosen for study, with two cut blocks being logged along each stream. One cut block near the headwaters of the stream will have a 30-m buffer strip (as per Alberta regulations) and a second, downstream cut block, will have no buffer strip. As with the lakes study, a number of limnological parameters will be evaluated with measurements made above, between and below treatments to determine differences. In both TROLS studies, 2 years of pre-harvest data will be collected in 1995 and 1996, harvesting will occur in the winter of 1996/97, and 3 years of post-harvest data will be collected in 1997–99. The stream study initially attempted to control and vary rate of harvest and spatial distribution within three replicated treatments. However, this design produced an excessive volume of timber that could not be handled by our mill. Following the current study, rate and spatial distribution of harvest will be incorporated as additional treatments.

Aquatic Ecosystem and Wildlife Management Under a Natural Disturbance Forestry Paradigm

Traditional sustained yield forestry involves a two- to three-pass system where a set of regularly shaped cut blocks are usually clear cut in a township over 1 to 2 years, followed by subsequent cuts in the same township spaced 15 to 20 years apart. Under this form of management, the patchwork of cut blocks left on the landscape, greatly reduces landscape variability, stand structure, shape and age distribution of the forest. Visual observation of the northeastern Alberta landscape strongly suggests

that current logging designs are far removed from the natural patterns of vegetation left by natural disturbances such as fire. We have just begun to study the spatial patterns of forest fires in our FMA with the intent of making our cut block shapes and sizes more closely approximate those of fire, the dominant form of natural disturbance in northeastern Alberta. Figure 1 shows the forest age structure along 19.3 km of a typical 60-m buffer strip along the Christina River in northeastern Alberta. Species composition and age structure within this riparian area is clearly variable, reflecting, in part, the variable nature of forest fires along water courses. How buffer strips may alter the variable nature of riparian vegetation, and how this links to aquatic ecosystems requires attention. Currently, forest fires in the boreal region of North America remove up to 2% of the forests each year. Our harvesting operations remove 0.3 to 0.5% of the forest each year.

We believe that by attempting to emulate fire patterns in our logging practices (e.g., irregular cut block sizes and shapes that follow topographic contours and natural vegetation boundaries), maintenance of terrestrial biodiversity (plants and animals) is likely much more attainable than under sustained yield forest practices. We also believe that the same principle may hold for aquatic ecosystems. Fire protection and the establishment of riparian buffers near streams may alter the fire return frequency along stream courses and the contribution of material (e.g., large woody debris) to the stream structure and function. Fire in riparian areas of boreal streams may function in resetting a stream's physical and chemical clocks. Conversely, buffer strips may inhibit the resetting of these clocks. Establishment of buffers along watercourses is a static management tool that seeks to maintain current conditions in waterbodies. Streams however, are highly dynamic entities. Fish communities that exist in a certain stream now were probably different 100 years ago. Streams constantly change, in terms of hydrology, channel shape, productivity, and especially large woody debris. How do buffer strips alter this dynamic process of change in streams? These questions require study.

Alberta-Pacific is currently collaborating with the National Hydrology Research Institute (Saskatoon, Saskatchewan) to investigate the impacts of a 130 000 ha forest fire that occurred in 1995 in the Mariana Lakes area of northeastern Alberta, on stream hydrology, water chemistry, and primary production of periphytic algal communities. The work will help develop the needed comparison between fire effects and logging. Fire is the baseline that logging should be compared to, not undisturbed ecosystems.

Traditionally, forestry-related activities have not incorporated a terrestrial or aquatic biodiversity component. Management of wildlife populations has focused on ungulates or other large mammals (Hunter 1991). While this fine-filter approach has been successful with some of the higher-profile wildlife species and for smaller, less visible species that share the same habitat, traditional wildlife management has limited our broader understanding of biodiversity. By focusing on maintenance of landscape patterns and processes under a natural disturbance model, Alberta-Pacific hopes to move from traditional wildlife management to a more holistic view based on biodiversity. This coarse-filter or ecosystem management approach assumes that by maintaining natural variability in landscape patterns, natural variation in animal communities will also be maintained (Hunter 1991; Salwasser 1991). Thus, the amount of material removed from a watershed (i.e., rate of extraction) and the spatial distribution of that material needs to be incorporated into both wildlife and aquatic ecosystem management.

Conclusions and Future Directions

Protection of aquatic resources in forestry has involved management by constraint through use of riparian buffer strips. This paper has outlined a new direction in forest management that Alberta-Pacific will investigate, namely, using a dynamic management strategy that operates on frequency of pulses into aquatic ecosystems through natural disturbance regimes. The goal of ecosystem management is to have harvesting effects fall within the natural range of variation caused by other disturbance agents (e.g., fire). To accomplish this, one needs to know the extent of variability in both terrestrial and aquatic ecosystems, and how fire effects compare to those caused by logging. For streams in particular, attention must also be directed toward other disturbance agents, including beaver activity and precipitation events, which greatly contribute to the natural variability in these systems. Clearly, many of the ideas expressed within this paper require scientific validation in order to develop sound management options. The land–water interface is a dynamic entity, for which broad generalizations across regions, provinces, or states cannot be made easily. As a starting point, Alberta-Pacific will examine and evaluate a new model of forest harvesting–a model based on natural disturbance. The move from clear cutting to

irregular shaped cut blocks with variable amounts of residual material is a first step towards developing forest practices that are ecologically sustainable. Future work could include variable landscape patterns, variable cutblock sizes, and the inclusion of prescribed fire.

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Riparian-in-a-Box: A Manager's Tool to Predict the Impacts of Riparian Management on Fish Habitat

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Abstract

"Riparian-in-a-Box" is an applied large woody debris (LWD) recruitment and channel function model. It allows resource managers to predict the effects of riparian forest management on site-specific, in-channel characteristics such as LWD abundance, and pool frequencies and areas. Model inputs describe the proposed riparian stand characteristics and the present channel conditions. Growth and mortality functions estimate future riparian conditions and a tree fall model approximates LWD recruitment to the channel. Empirical relationships between LWD abundance and pool abundance predict in-channel conditions. With this model, managers can evaluate how effectively different riparian management options achieve specific in-stream goals. Model results for the Griffin and Tokul watersheds in King County, Washington demonstrate that different sized streams respond dissimilarly to riparian management. In-channel responses were compared for variously sized streams with thinned and unthinned buffers. The thinning treatment consisted of removing the smaller trees, releasing growing space to larger trees. Pool surface areas of smaller streams increased more quickly with unthinned buffers, while larger streams showed increased benefit after riparian thinning.

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Introduction

Foresters and fish habitat biologists in western North America recognize that woody debris creates physical fish habitat in stream channels (Bisson et al. 1987; Trotter 1995), and that riparian and floodplain forests are the primary source of large woody debris (LWD) recruitment to stream channels (Sedell and Frogatt 1984; Bisson et al. 1987; Murphy and Koski 1989; Montgomery et al. 1995). Land management activities that remove trees within a riparian zone alter the density and size distribution of recruitable trees and affect stream habitat characteristics such as pool abundance, in-channel cover, and sediment retention structures (Potts et al. 1990; Bilby and Ward 1991). A change in habitat can affect fish utilization in a particular stream reach because different fish species have specific habitat preferences at different life stages (e.g., Bisson et al. 1988). Although these different research elements can be conceptually linked to understand the influences of riparian forests management on habitat features, quantifying habitat changes caused by a change in large woody debris recruitment from riparian management activities has proven more difficult.

When government agencies regulate land use actions within riparian areas, they must balance fish habitat protection or restoration with other economic considerations. In this context, effective management of riparian forests and fish habitat requires accurate estimates of both current and future LWD inputs, as well as of stream responses to different riparian management strategies. This paper presents a model that was developed in response to this need: Riparian-in-a-Box. The purpose of the model is to predict the effects of different riparian buffer treatments on measurable features of channel morphology, particularly pool surface areas. These model results can then be compared to agreed upon in-stream management objectives, and, if the procedure is used within a decision process with a feed-back loop, monitoring of stream responses provides information to improve future modeling efforts (Walters 1986; Oliver et al. 1992; Bisson and Raphael 1996). In this paper, we present the model and discuss management implications for the Griffin and Tokul watersheds in western Washington.

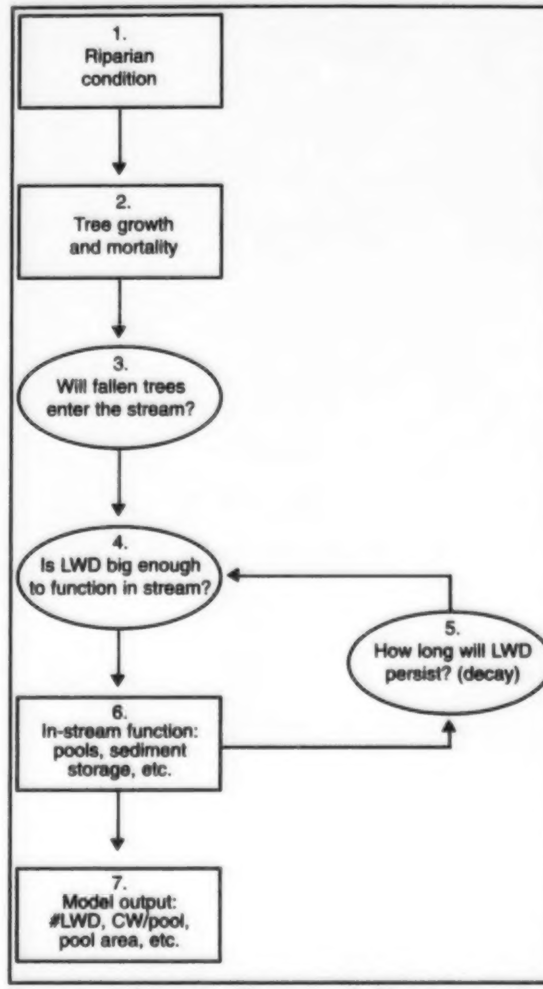

Figure 1. Schematic diagram of Riparian-in-a-Box model structure.

Description of The Model

Riparian-in-a-Box was designed to model growth and mortality of riparian forests, LWD recruitment to a stream, and effects of changing LWD abundance on pool frequency and areas (Fig. 1). It consists of two separate models, one that grows trees and estimates mortality (Boxes 1 and 2), and one that estimates tree fall, entry to a stream, and LWD function (Boxes 3 through 7). Given the current condition of a riparian stand and adjacent stream channel, the model calculates future tree growth and mortality. When a tree dies and falls, the model determines if the tree will enter the stream, and if it is large enough to influence channel morphology. At each time step, a feed-back loop is used to evaluate the persistence of in-channel LWD. Finally, the amount of functional LWD is determined, and pool numbers and pool surface areas are predicted.

The model was developed for applications to Pacific Northwest streams in managed forests. It can be applied to riparian stands that consist primarily of Douglas-fir (*Pseudotsuga menziesii*), and to streams

with bankfull channel widths from 5 to 30 m and gradients of 6% or less. The model riparian zone has an arbitrary length along a stream of fixed bankfull width, and the maximum riparian width is equal to the height of the tallest trees in the zone. For computation purposes, the riparian stand is partitioned into discrete bands parallel to the channel with uniform characteristics within each band. Stand attributes may vary among bands.

Like previous modeling efforts (e.g., Van Sickle and Gregory 1990), Riparian-in-a-Box assumes that LWD inputs consist of whole trees falling directly into a stream channel from an adjacent hillslope, terrace, or floodplain (Table 1). Input of tree parts (branches and broken crowns) is not considered. Other LWD delivery processes not accounted for include: 1) LWD transported into a reach from upstream by fluvial means; 2) LWD delivered by mass wasting; and 3) LWD that slides or rolls downhill before entering a stream. The relative significance of these LWD delivery mechanisms, and the implications of these assumptions are discussed in Van Sickle and Gregory (1990).

The initial condition of the riparian vegetation (Box 1 in Fig. 1) in each stand is characterized by: 1) tree species (Douglas-fir only in this case study); 2) tree density (trees/area); and 3) frequency distributions of diameters at breast height (DBH) and tree heights. These data are input into the tree growth model. In the pilot application of Riparian-in-a-Box, a proprietary growth model (PCFIR) developed by the Weyerhaeuser Company, Tacoma, Washington was used (Box 2 in Fig. 1). It simulates both tree growth and mortality, and includes effects of shading understory trees by dominant trees. The model applies to Douglas fir trees in managed forests, and simulates trees to 200 years of age (N. Vogt, Weyerhauser Co., Tacoma, Washington, personal communication). Growth and mortality by suppression are calculated based on empirical data, and growth rates are controlled by the initial tree stocking levels, the site's growth potential index, and whether or not fertilizer is applied. Tree taper is not considered and the boles are considered to be cylindrical (Table 1).

In addition to loss of trees from overstory shading, average mortality from windfall is considered. The proportion of initial trees in the stand that have fallen (P_F) after time (t) is calculated by the equation:

(1) $P_F = (1-(1-T_F)^t)$

where T_F = tree fall rate. After harvest of the adjacent stand, the following tree fall rates are assumed: 20% for the first decade, 15% for the second decade, and 10% equilibrium rate for all following decades. The initial rate is derived from a study by Andrus and Froehlich (unpublished report) on windfall losses in the first decade after harvest on streams in coastal Oregon. The equilibrium rate after 20 years is based on research by Murphy and Koski (1989) in southeast Alaska. The rate for 10 to 20 years was arbitrarily set between the other two values.

While windthrow is the primary cause of tree loss in riparian buffer stands left adjacent to harvested areas (Steinblums et al. 1984), there are other riparian processes not considered. Loss of trees by

Table 1. Assumptions used in development of the Riparian-in-a-Box model

Model components	Location in Figure 1	Assumptions
Riparian condition	Box 1	• one species—Douglas-fir • wood delivery by direct tree fall only
Tree growth and mortality	Box 2	• mortality from competition and windfall only
Tree fall	Box 3	• trees fall independently of each other • random tree fall direction • no breakage during tree fall • trees do not span channel
Functional wood size	Box 4	• trees with and without root wads act similarly
Debris volumes and pool areas	Box 7	• no tree taper • averaged tree volume enters channel

bank erosion and flooding by the stream is not explicitly considered, although some amount of tree removal from these methods is implicit in the tree fall rate source data. Other sources of mortality, such as losses from disease, fire, and insect infestations are also not considered.

After a tree falls, a geometric model is used to determine what portion of the tree, if any, enters the channel (Box 3 in Fig. 1). Diameter distributions of LWD delivered to the channel are calculated using a probabilistic formulation (Van Sickle and Gregory 1990). The model assumes: 1) trees fall independently of each other; 2) tree fall direction is random; 3) trees do not break during the fall; and 4) fallen trees do not span, but enter V-notched channels (Table 1). These assumptions imply that neither tree lean nor steepness of the riparian slope influence fall direction, and there is no preferred wind direction causing wind throw.

The probability of a falling tree (PS) entering a stream under these conditions is:

(2) $P_S = \cos^{-1}(z/h)/\Pi$

where z = the perpendicular downslope distance from a standing tree to the nearest channel boundary and h = tree height (Van Sickle and Gregory 1990). The number of trees available for recruitment is simply the density of trees (D) in a particular band of the riparian zone, multiplied by the length (L) and width (W) of that band. Then, for each band of the riparian buffer with uniform characteristics the total number of fallen trees delivered to a stream (N_I) is:

(3) $N_I = D(L)(W)P_F\,P_S$

where P_F = probability of a given tree falling after a time interval (equation 1), and P_S = probability of a falling tree entering the stream (equation 2). The totals for each riparian band are then summed together to give the total number of trees entering the stream for the entire riparian buffer.

The tree fall model can predict total volumes of delivered LWD when applied to each individual tree in the model riparian buffer and the results are summed. However, we calculated delivered LWD volume in a simpler fashion. We first calculated the average volume of a functional LWD piece for a given stream channel width (using the average DBH and height of functionally sized LWD), then multiplied that average volume by the number of functionally sized pieces that enter the stream during a time step. This method yields a result similar to that which could be calculated by individual trees, yet it overestimates volumes because of two assumptions: (1) trees are not tapered, and (2) the entire tree enters the channel.

To this point, the model has predicted the delivery of LWD to a channel from the riparian buffer. Next, the model determines if the LWD is large enough to function in the channel by comparing the DBH of a fallen tree with the minimum functioning size of LWD (Box 4 in Fig. 1). It then tallies functionally sized LWD added to the channel during each time step. The minimum functioning size of LWD increases linearly with channel width (Fig. 2) by the following equation:

(4) DBH = 3.06(CW) + 22.10

where DBH = diameter breast height of LWD in centimeters and CW = channel width at bankfull flows in meters. The regression is based on data from western Washington (Bilby and Ward 1989; DNR 1996; G. Pess, Tulalip Tribes natural Resources Department, Marysville, Washington, unpublished data).

The model also calculates the amount of LWD lost over time (Box 5 in Fig. 1). The depletion processes considered in the model include decay of trees (until its size is smaller than the functional threshold) and net loss of functionally sized woody debris during floods. One study conducted in southeast Alaska found depletion rates for old-growth conifer varied from 14 to 20% per decade, as a function of both LWD size and channel size (Murphy and Koski 1989). Considering only channels up to 20 m wide and only functionally sized LWD, the range collapses to 14 to 15%. Therefore, we used a depletion rate of 15% per decade. Depletion of LWD over time is calculated with an equation of the form $y = (1-x)^t$, where x = depletion rate per year and t = elapsed time in years. The basic model output is the number of functional LWD pieces after depletion, as a function of time. These LWD numbers can be converted to a variety of formats including number of functional pieces per channel width or length.

Only a certain percentage of functionally sized LWD actually creates pools (Box 6 in Fig. 1). Data from western Washington indicate that 3 of 10 pieces of functionally sized LWD create pools (Montgomery et al. 1995; DNR 1996). Hence, we used a recruitment factor (R) of 0.3. Of the pools formed by LWD, a portion are from single pieces of LWD, and the remainder are from debris jams with more than one piece of functionally sized LWD. Based on data from northwestern Washington state, USA (DNR 1996), we found that 35% of all pools were associated with debris jams (J = 0.35), and 25% with single pieces of LWD (L = 0.25), and, on average, two pieces of functionally sized LWD (K = 2) were needed in order for a debris jam to form.

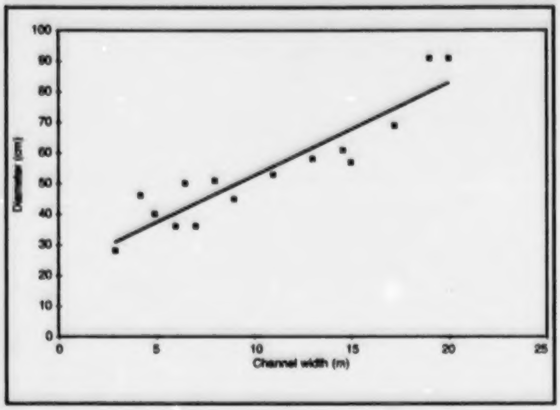

Figure 2. Size of functional LWD as a function of channel width. Each data point represents the average of hundreds of individual measurements for a single stream reach. No distinction was made between LWD with or without root wads.

From these data, we developed a formula to estimate pool spacing from a number of functionally sized LWD (N_f):

(5) $G = (1/N_f)(L/CW)(1/R)[(J)K + S]$

where G = number of channel widths (CW) per pool, L = length of stream segment, CW = bankfull channel width, R = recruitment factor, J = proportion of pools from debris jams, S = proportion of single LWD pools, K = number of functional LWD pieces in a debris jams. For example, a stream segment 15 metres wide and 100 metres long with 15.5 pieces of functionally sized LWD, has a pool spacing of 2 channel widths per pool.

Pool surface areas are calculated based on debris volumes (DV) and bankfull channel widths (Table 3). These equations are used to estimate pool surface areas at each time step. The results may be presented as absolute numbers (e.g., total pool area in a reach through time), the percentage change in a value through time, or the cumulative change in a value through time.

Model Application Case Study

Riparian-in-a-Box was used to evaluate different riparian proposals for the 170 km^2 Griffin and Tokul watersheds 30 km east of Seattle, Washington. The watersheds produce an average of 20% of the total coho salmon for the Snohomish River, though comprising only 3% of the total watershed area. Large woody debris removal from streams and insufficient riparian buffers left by past management have reduced in-stream LWD and converted stream segments that were previously pool/riffle sequences into predominantly long riffles. The present management goal was to transform plane-bed segments (based on the channel classification scheme of Montgomery et al. 1993) into forced pool/riffle sequences. The specific objective was to identify the riparian treatment that would most quickly restore and maintain desired pool surface areas through time. Pool surface area was specifically chosen because it is the pool attribute that most strongly influences coho habitat production potential (Reeves et al. 1989).

Thinned and unthinned buffers were modeled on plane-bed stream segments with bankfull widths varying from 5 m to 30 m, and channel gradients 6% and less. In all thinning cases, only the smallest trees are removed to accelerate growth of the remaining stand, and thinning was excluded within 9.1 meters of the stream. Initial stand ages, tree densities, and buffer widths of the treatments considered are summarized in Table 2.

Results from one thinning treatment of a fifty-year-old riparian stand on three different channel widths were projected 150 years into the future (resulting in 200-year-old trees in the stand). Of greatest interest was the cumulative change in percent pool area over that time period (Fig. 3). The three thinning cases were compared to the unthinned case, represented by the plane on the graph associated with the 0% cumulative change in pool area. Buffer strips thinned to 148 trees/ha on narrower streams (e.g., 4.6 m wide) result in less pool surface area through time than unthinned buffer strips. However, wider streams (e.g., 10.4 m width) have increased pool surface areas compared to the unthinned case. The bankfull stream width at which thinning begins to positively affect pool surface area (hereafter referred as the cross-over width) is approximately 7 m in this case. When buffers are thinned less intensely, resulting in a higher initial tree density, the cross-over width increases. For example, at a post thinning density of 346 trees/ha, only streams wider than approximately 10.5 m have elevated pool surface areas compared to the unthinned scenario.

Sensitivity analysis and error

Significant potential error is introduced by several modeling assumptions (Table 1). Additional

Table 2. Treatment scenarios for model runs used in the Griffin-Tokul case study

Treatment	Initial tree age	Distance from stream	Tree density	Fertilizer applied
Unthinned	50 years	0–30.5 m	556 trees/ha	no
Thinned	50 years	0–9.1 m	556 trees/ha	no
		9.1–21.5 m	multiple cases with 148 to 346 trees/ha	yes and no

Table 3. Regression equations used to predict pool areas based on debris volume and channel width (Bilby and Ward 1989). PA = pool surface area in m²; DV = debris volume in m³; CW = channel width in m.

Channel width range	Regression equation
< 7 m wide	$\log_{10}PA = 0.38 \log_{10}DV - 2.61 \log_{10}CW + 2.24$
7–10 m wide	$\log_{10}PA = 0.64 \log_{10}DV + 0.49$
> 10 m wide	$\log_{10}PA = 0.64 \log_{10}DV + 1.31 \log_{10}CW - 0.77$

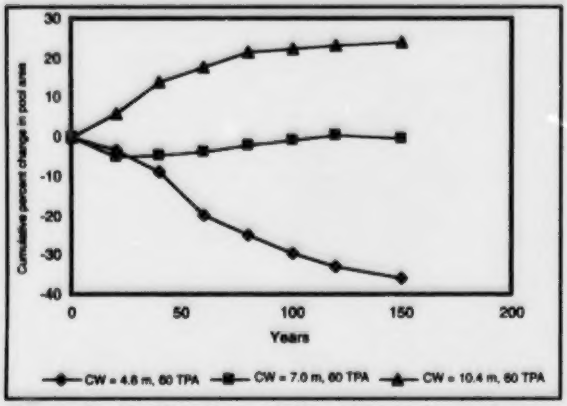

Figure 3. Predicted cumulative change in pool area over time for the same thinning treatment on three channels of different widths. Percent change is measured relative to the predicted pool area for an unthinned riparian zone.

uncertainty is associated with model parameters, particularly rate of tree growth (Box 2 in Fig. 1), tree mortality (Boxes 2 and 3 in Fig. 1), and in-stream depletion (Box 5 in Fig. 1). Here we consider two types of error: absolute and relative. Absolute error reflects differences between predicted values and actual values, whereas relative error reflects changes in the relative ranking of different modeling scenarios in terms of a selected output. That is, relative error is the change in relative ranking of model outputs as input parameters are varied. We considered large relative errors to be those which change the ranking of scenarios in terms of the amount of functionally-sized LWD present in a channel after a specified number of years.

Absolute and relative results are most sensitive to model inputs from the proprietary tree growth model PCFIR (Weyerhauser Co., Tacoma, Washington). The sensitive inputs from the tree growth model include, in decreasing order of significance, initial riparian condition (tree densities and sizes), tree growth rate, and suppression mortality (Weyerhaeuser 1996). For these parameters we did not perform a specific sensitivity analysis. Rather we inferred the sensitivity to these parameters from changes in the total number of pieces of functioning LWD over time during initial trials with the proprietary PCFIR growth model. The model appears to be most sensitive to growth model parameters because small differences in these inputs cause increasingly large changes in number of functioning LWD over time.

For parameters specific to the recruitment model (treefall rate and instream-LWD depletion rate), we performed a sensitivity analysis to determine their relative influence on model outputs. We varied the model parameters over the range of values expected on managed forests, and considered the effect of varying parameters on both the absolute and relative results of two model output types: total number of pieces of functioning LWD and cumulative change in pool area. Number of functioning pieces of LWD after 150 years is more sensitive to in-stream LWD depletion rates than treefall rates (Fig. 4). Because pool area is a linear function of number of functioning LWD, the sensitivity plot for cumulative change in pool area after 150 years appears virtually identical to Figure 4.

The combination of error sources and the sensitivity of the model to expected variations in model

parameters lowers our confidence level in the absolute model outputs. For example, by varying depletion rate from 5 to 15% per decade the values of predicted number of pieces of functional LWD for an otherwise similar stream segment vary by more than 100%. However, we have greater confidence in the relative results. To test the accuracy of the relative results, we varied both tree fall and in-stream depletion rates, and compared the LWD and pool surface area outputs. Varying tree fall rates from 5 to 25% and in-stream depletion rates from 15 to 10% had no influence of the relative LWD outputs. This held true when the parameters were varied in ways to maximize changes in model output (e.g., a high tree fall rate with a low in-stream depletion rate). Relative pool surface area changes were not altered by variable tree fall rates and were insignificantly affected by fluctuations in in-stream depletion rate. However, at channel widths near the cross over width (6.2 to 7.6 m in this example), the influence of the thinned or unthinned treatments varied as model input parameters were varied. That is, the thinned scenario produced a slight increase in pool surface areas relative to the unthinned case, but the essential results were unaffected.

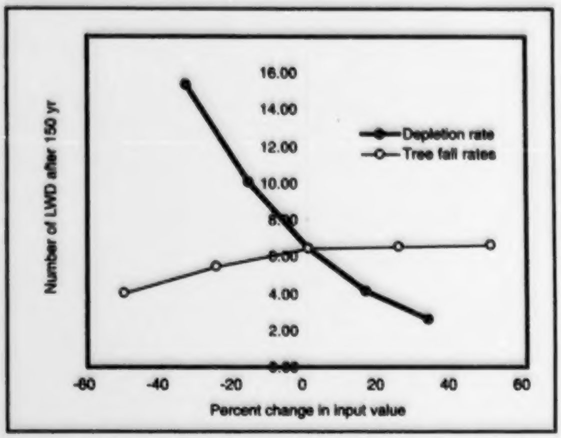

Figure 4. Sensitivity of number of pieces of LWD after a 150-year modeling period to changes in in-stream depletion rate and treefall rate. Each parameter was varied over its natural range with 0% change representing a mid-range value. Mid-range long-term treefall rate was 0.10 per decade; mid-range depletion rate was 0.15 per decade.

Discussion

We have developed a model that predicts the effects of different riparian treatments on site-specific, in-channel characteristics related to fish habitat. The model utilizes site-specific and regional data to predict relationships between riparian stand characteristics, LWD abundance, and pool spacing and area. The model makes it possible to unambiguously compare the influence of different riparian treatments on measurable channel attributes. The model has been used in a variety of existing management processes including the prescription phase of Washington state watershed analysis and development of Habitat Conservation Plans under the USA *Endangered Species Act*. It can also be used to compare the effects of different regulatory strategies for riparian forest management.

Stream size is a crucial attribute to consider when designing riparian buffers. Model results show that smaller streams respond differently to thinning than larger streams. For all streams, understory thinning results in fewer but larger trees in the riparian buffer over time. Because smaller LWD functions in smaller streams, these streams are more sensitive to increased number of LWD than to increased size of

Table 4. Expected improvements to the Riparian-in-a-Box model

Model component	Current version	Future version
Tree growth model	• proprietary • Douglas-fir only • no tree stem taper	• public domain • multiple species and canopy heights • includes tree stem taper
Tree fall model	• calculations on averaged classes of trees • each tree modeled separately	• averaged tree volume enters channel • individual in-stream tree volumes calculated
LWD functions	• proportion of LWD in log jams is constant • static LWD-pool relationship • no sediment storage function	• LWD in log jams varies with channel size and LWD loadings • LWD-pool relationship varies with gradient • includes sediment storage

LWD. Hence, small streams show little benefit from understory thinning. Larger streams require larger LWD to function, and are more responsive to larger LWD. Therefore, larger streams benefit from understory thinning because the functional LWD size is reached in a shorter period of time. This pattern holds true for both the pool spacing response to number of LWD and the pool surface area response to more and larger LWD.

Development continues on the model to improve accuracy and extend the range of applicability (Table 4). The new model will be used to determine the influence of the following factors on in-stream LWD distribution and channel function: 1) riparian disturbance (both natural and anthropogenic); 2) riparian composition (including vegetation class and seral stage); and 3) stream channel type. Additionally, we will explicitly consider the effect of in-stream LWD on coarse sediment storage, and will relate alterations in channel function to changes in fish community composition.

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Sediment Generation from Forestry Operations and Associated Effects on Aquatic Ecosystems

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Abstract

Timber harvest operations have been shown to have many effects on adjacent watercourses and on the aquatic ecosystems they support. This may occur from introductions or loss of woody debris, loss of riparian vegetation, accelerated stream bank and bed erosion, the alteration of natural channel form and process, and the reduction of stream habitat diversity. However, the existing literature indicates one of the most insidious effects of logging is the elevation of sediment loads and increased sedimentation within the drainage basin.

Sediment generation from various forestry practices has been studied extensively in the past. Forestry practices that generate suspended sediments include all operations that disturb soil surfaces such as site preparations, clear-cutting, log skidding, yarding, slash burns, heavy equipment operation and road construction and maintenance. From these sources, construction, use and maintenance of logging roads located near watercourses produce by far the highest levels of suspended sediment generation in streams.

Three aspects of logging road development and maintenance are known to elevate sediment loads in watercourses: 1) instream and near-stream construction operations; 2) reduction in retention time and associated increase in erosion in the drainage basin; and 3) mass soil movements and/or landslides associated with logging road design and placement.

This literature review examined the effects of increased sediment load and sedimentation on aquatic ecosystems emphasizing forestry operations that generate elevated sediment loads. The review included the effects of sediment on fish (behavioral, physiological and population effects) and the effects of sedimentation on fish habitats (including spawning, rearing, food production, summer and overwintering habitats). A habitat effects relationship was presented that related the concentration and duration of specific sediment exposure events to the alteration of fish habitats. This relationship allows for post-disturbance evaluation of the potential effects on fish habitat.

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Many advances have been made in the design and construction of logging roads over the past three decades. Many of these advances have been made in the design of road access systems, while other advances have been made in the development of more sophisticated mitigation techniques. These advances are described and discussed in relation to minimizing sediment generation and subsequent sedimentation in aquatic environments.

Introduction

Forest harvesting practices can elicit a number of physical changes within a watershed. These changes can set up associated responses in a wide range of physical, chemical and biological processes and can substantially alter aquatic habitats and communities. Forestry-related activities can influence: stream hydrologic regime by reducing the time between peak rainfall and peak stream discharge and consequently increasing the magnitude of peak seasonal flows; water quality parameters such as temperature and sediment load; and, stream geometry by increasing erosive forces, channel migration, width to depth ratio and altering stream form and process (Meehan 1991; Salo and Cundy 1987).

Forestry-related activities are not always harmful to freshwater fish communities. Long-term and intensive study of logging on fish communities of Carnation Creek, Vancouver Island, British Columbia, revealed the annual growth of juvenile coho salmon (*Oncorhynchus kisutch*) increased in years immediately following logging, which was likely in response to increases in air and water temperatures (Hartman and Scrivener 1990; Holtby 1988; Tschaplinski and Hartman 1983). However, forestry activities have been shown to have adverse effects on resident and migratory fish communities, often related to an increase in the delivery of sediments to streams.

Although frequently viewed as synonymous, suspended sediments and sedimentation are two discrete processes that affect aquatic communities in different ways. Sediment transport in watercourses occurs in three forms, which include wash load, suspended, load and bed load. The entry of sediments to watercourses from upland sources is a natural process; however, when the rate entering a watercourse exceeds the capacity of the watercourse to transport or assimilate the sediment, stress may occur to the aquatic community and the suitability and/or productivity of aquatic habitats may be altered. Excess sediment in rivers and streams has been identified as the largest and most pervasive water pollution problem faced by aquatic systems in North America (Sweeten 1995).

This paper provides a collection of information related to biological responses to sediment synthesized from the literature and attempts to uncover relationships that are present between elevated sediment episodes and biological response. For additional information pertaining to this topic or to obtain additional references, the reader is directed to Meehan 1991, Salo and Cundy 1987, Newcombe (1994), Newcombe and Jensen (1995), Kerr (1995), Waters (1995) and Anderson et al. (1996).

Sources of Elevated Sediment Loads

Anthropocentric Sources of Elevated Sediments

Erosion within streams is a natural process and is affected by parameters such as stream flow, channel structure and stability, streambed composition, and disturbance within the watershed such as fire, landslide or ice scour. The disturbance of lands accelerates erosion and increases the delivery of sediment to stream systems. Any activity that is undertaken within a watershed that disturbs land surfaces has the potential to increase sediment delivery to streams. Most human activities will increase erosion to some extent in forested watersheds. Human activities that most frequently increase sediment loads in watercourses include agriculture, mining, forestry, urban development, and stream channel alteration such as dams and channelization, and instream construction associated with developments such as bridges, roads or pipeline and transmission crossings. This paper discusses the effects of sediment increase associated with episodic elevated sediment events in general; however, types of mitigation to minimize sediment load increases, which are specific to the forestry industry are discussed in the Mitigation Measures section and sediment sources specific to forest harvest operations are discussed below.

Forestry Sediment Sources

Forestry operations frequently increase sediment delivery into streams. Logging operations have been shown to increase sediment production above natural sedimentation rates (Megahan and Kidd 1972).

The activities that commonly result in increased sediment delivery include clear-cutting, skidding, yarding, site preparation for replanting and road construction, and use and maintenance (Waters 1995).

Among forest harvesting activities, disturbance associated with logging road construction and operation produces the greatest sediment load increase (Waters 1995; Furniss et al. 1991; Cederholm et al. 1981). Roads associated with a jammer logging system in the Payette National Forest, Idaho, increased sediment production an average of about 750 times over the natural rate over a 6-year period following construction (Megahan and Kidd 1972). The average erosion rate from roads on the jammer unit for 1.35 years preceding logging was 56.2 tons/mile^2/day, and the average rate for 4.8 years following logging was 51.0 tons/mile^2/day (Megahan and Kidd 1972).

Effects of Elevated Sediment Loads

Sediment Effects on Fish

In response to changes in the environment, ecosystems often undergo changes in community composition and structure. Organisms respond to environmental change in order to avoid or minimize effects on fitness. If an organism can not compensate for a change in the environment and suffers a reduction in fitness, the environmental change is termed a stress (Brett 1958; Koehn and Bayne 1989). Therefore, a stress limits either the rate of resource acquisition or growth and reproduction so that fitness is reduced (Grime 1989).

Stress has been defined as "the sum of all the physiological responses by which an animal tries to maintain or re-establish a normal metabolism in the face of a physical or chemical force" (Selye 1950). Stress occurs when the homeostatic or stabilizing processes of the fish or organism are extended beyond the capabilities of the organism to compensate for the biotic or abiotic challenges. Anthropogenic inputs of sediments into stream and riverine environments can cause stress to aquatic systems and thereby directly and indirectly impact upon fish behavior and health. Increased concentrations of suspended sediments can have direct effects on fish behavior, fish physiology and fish populations (Anderson et al. 1996).

Behavioral Effects

Changes in fish behavior are some of the first effects evoked from increasing concentrations of suspended sediments. Behavioral changes are generally considered benign and transitory. They are easily reversed and do not exhibit a long-lasting impact (Newcombe 1994). Typical responses include an increased frequency of the cough reflex, avoidance of suspended sediments, reduction in feeding and temporary disruption of territoriality. The severity of the behavioral response is associated with the timing of disturbance, the level of stress (and associated energy cost) and the importance of the habitat from which the fish may be being excluded.

The avoidance of suspended sediment plumes is one of the first reactions. Bisson and Bilby (1982) observed this behavior evoked in juvenile coho salmon at total suspended sediment (TSS) concentrations as low as 88 mg/L. Similar results were recorded by McLeay et al. (1987) who found that Arctic grayling (*Thymallus arcticus*) avoided concentrations greater than 100 mg/L.

Increased concentrations of suspended sediment have also been correlated with a reduction in feeding. Feeding rate may be a function of prey visibility. McLeay et al. (1987) states that Arctic grayling exposed to suspended sediment concentrations greater than 100 mg/L were slower to recognize the food and more frequently missed a food item when they attempted to eat it. Sigler et al. (1984) believes, however, that reduced feeding is more complex than reduced ability to see prey items, as many fish species (especially benthic feeders) do not use sight to identify prey items, but still exhibit reduced levels of feeding in response to elevated sediment loads.

High concentration of suspended sediments has also been associated with the loss of territoriality and interruption of movements of salmonids. Berg and Northcote (1985) found that territorial behavior was lost at concentrations exceeding 30 NTU. They indicate that territoriality is influential in the allocation of food and habitat resources. Disruption to territoriality can occur when turbidity limits the visual distance that individuals can see, and when the downstream drift of fish avoiding increased concentrations of suspended sediment disrupts existing territories.

Physiological Effects

Physiological changes can be measured in fish as a response to the increased stress of suspended sediments. The typical measured responses include impaired growth, histological changes to gill tissue, alterations in blood chemistry, and an overall decrease in health and resistance to parasitism and disease. The effects of sediment exposure on each of these physiological effects are discussed below. When compared to sediment exposure events that

elicit behavioral responses, longer exposure periods and/or higher concentrations are generally required before physiological responses are expressed. In this respect, physiological responses are more of a chronic effect. The effects are usually a graded response to increasing sediment dose. Impacts evoked from lower doses can be transitory, while those resulting from higher doses can be more lasting and severe.

Growth

An impaired growth rate is generally one of the more sensitive physiological responses to an increase in suspended sediment concentration. Unlike behavioral responses, impaired growth generally requires a longer exposure period before effects are manifested. Sigler et al. (1984) found that growth was impaired in juvenile steelhead trout (*Oncorhynchus mykiss*) and coho salmon exposed to fire clay or bentonite clay at concentrations between 84 and 120 mg/L during a 14 to 21 day exposure period. Similar concentrations of 100 mg/L or greater were found to significantly impair growth in Arctic grayling under-yearlings (McLeay et al. 1987), largemouth bass (*Micropterus salmoides*), bluegill (*Lepomis macrochirus*), and redear sunfish (*Lepomis gibbosus*) (Buck 1956). However, growth impairment may be related more to the metabolic demands resulting from stress caused by increased suspended sediment than from a reduction in feeding. The time required before growth impairment was measurable ranged from a low of 2 weeks for juvenile steelhead trout and coho salmon to a high of 6 weeks for Arctic grayling under-yearlings (Sigler et al. 1984; McLeay et al. 1987).

Blood Chemistry

Alteration in blood chemistry resulting from the increased stress of suspended sediments has been found to be associated with concentrations ranging between 500 and 1500 mg/L (Redding and Schreck 1980; Servizi and Martens 1987). The changes most commonly recorded include an increase in haematocrite, erythrocyte count, hemoglobin concentration, and elevated blood sugar levels (hyperglycemia), plus decreases in blood chloride content, and depletion of liver glycogen (Wedemeyer et al. 1990; Servizi and Martens 1987). These increases coincide with the release of stress hormones (i.e., cortisol and epinephrine) and traumatization of the gill, and presumably represent a compensatory response to a decrease in gill function (Newcombe 1994). In addition, Sherk et al. (1973) found these changes to be associated with a reduction in the swimming endurance of white perch (*Morone americana*) exposed to 650 mg/L of TSS. Most of the observed changes resulted after four to five days of exposure (Newcombe 1994). Exceptions to this, however, were noticed by Redding and Schreck (1980) who found a significant increase in haematocrit volume within steelhead trout after only 9 hours of exposure to 500 mg/L of volcanic ash, clay and topsoil.

Gill Trauma

Increased concentration of suspended sediments are known to physically traumatize gill tissue. The primary mechanisms of action is through physical abrasion of tissue and particle adsorption onto the gill. The types of tissue changes observed include swelling of secondary lamella and hypertrophy (cell swelling) of epithelial cells (Sherk et al. 1973); hyperplasia (increase in cell number) of gill tissue (Simmons 1984); and tissue necrosis (Servizi and Martens 1987).

The severity of damage appears to be related to the dose of exposure, as well as the size and angularity of the particles involved. Greater damage is typically observed with larger, more angular particles (Servizi and Martens 1991). These factors could account for the large range in responses seen for different exposure rates. For example, concentrations as low as 270 mg/L are known to cause gill damage in rainbow trout (*Oncorhynchus mykiss*) (after 13 days of exposure (Herbert and Merkens 1961) and yet McLeay et al. (1987) found no gill damage in young-of-the-year (YOY) Arctic grayling that were exposed to concentrations as high as 1300 mg/L; the duration of exposure was, however, only 4 days.

Secondary effects resulting from an infestation of parasitic protozoans were found in juvenile rainbow trout that were exposed to extremely high concentrations of suspended sediments. The trout were exposed to 4887 mg/L for a period of 16 days (Goldes 1983). This author did note that the protozoan infection and gill architecture was found to be normal 58 days after the exposure ceased.

Resistance

Increased concentrations of suspended sediments have been associated with an overall decrease in the ability to defend against disease and to tolerate chemical toxins. For example, Herbert and Merken (1961) observed rainbow trout to be more susceptible to infestations of fin rot when fish were exposed for 121 days to concentrations of 270 mg/L of diatomaceous earth. Likewise, Servizi and Martens (1991) found a correlation between the

prevalence of a viral kidney infection and an increased concentration of suspended sediments in coho salmon. When concentrations of suspended sediments exceeded 100 mg/L, the tolerance of Arctic grayling to the toxicant pentachlorophenol (PCP) decreased (McLeay et al. 1987). This observation by McLeay et al. (1987) indicates a general decrease in tolerance to increased environmental stressors.

Phagocytosis

A process that may be closely linked to reduced resistance, is phagocytosis. Newcombe and Jensen (1995) discuss the process by which fine particles are enveloped by cells within fish gill and gut tissues and are transported to internal repository tissues. The main organ of repository in fish is the spleen (Newcombe and Jensen 1995). It is hypothesized that through this process, particles could reduce resistance to other stressors by impairing fish health. In addition, particles could trigger tumour induction, especially in circumstances where contaminants were adsorbed to particles in suspension (Newcombe and Jensen 1995).

Lethal Effects

Increased concentrations of suspended sediments and increased sedimentation rates have the potential to affect fish populations. The primary mechanisms of action are through increased egg mortality, reduced egg hatch, a reduction in the successful emergence of larvae, plus the sediment-induced death of juvenile and adult fish. These mechanisms are discussed below.

Egg Mortality

The primary cause of egg death is generally from burial by settled particles. Thin coverings (a few milimetres) of fine particles are believed to disrupt the normal exchange of gases and metabolic wastes between the egg and water. Sedimentation rates of 0.03 to 0.14 g dry weight sediment/cm^2 (i.e., 1–4 mm depth of silt and clay) significantly reduced the survival of lake whitefish (*Coregonus clupeaformis*) eggs (Fudge and Bodaly 1984). The effects upon egg mortality appear to be more closely related to the sedimentation of particles and less related to the concentration of suspended sediments. Zallen (1931) observed that concentrations of 1000 to 3000 mg/L had no effect upon the survival of mountain whitefish eggs (*Prosopium williamsoni*). Campbell (1954), however, found 100% mortality in rainbow trout eggs exposed to TSS concentrations of 1000 to 2500 mg/L. Differences in egg mortality effects associated with elevated sediment loads is related to the size of the sediment particles involved and rates of sediment deposition.

In addition to the concentration of suspended sediments and the size of the particles involved, the duration of exposure appears to be a key factor in determining the effects of sediments on egg survival. Slaney et al. (1977) noticed that hatching success for rainbow trout was reduced after 2 months of exposure to 57 mg/L. A significant reduction in the hatching success of white perch and striped bass (*Morone saxatilis*) was observed in only 7 days after exposure to about 1000 mg/L TSS (Auld and Schubel 1978). The magnitude of the effect of sediment exposure may also be influenced by the timing of sediment exposure with respect to the stage of embryo development. The dose of sediment required to induce egg mortality is greatly influenced by the physical characteristics of the stream which, in turn, affect sediment transport capabilities and the capacity to maintain sediments in suspension or otherwise to result in their deposition.

Juvenile and Adult Fish Death

Juvenile and adult fish generally appear to be more resilient to stress from suspended sediments than other life history stages. Short-term increases in TSS concentrations between 11000 and 55000 mg/L appear to be the point at which salmonid mortality significantly increases (Stober et al. 1981; Servizi and Martens 1987; Smith 1940). McLeay et al. (1983) reported survival of Arctic grayling subjected to moderately high concentrations (1000 mg/L) of fine grained materials (mining silt). Lloyd (1987), in a review of existing information, reported lethal effects to fish at concentrations ranging from 500 to 6000 mg/L. Sigler et al. (1984) reported mortality in young of the year coho salmon and steelhead trout at 500 to 1500 mg/L. Based on the information available on sediment and acute effects to fish, it is apparent that the severity of effect caused is a function of many factors which, in addition to concentration, duration particle size and life history stage, may include temperature, physical and chemical characteristics of the particles, associated toxicants, acclimitization, other stressors and interactions of these and other factors (Waters 1995).

Habitat Effects

Habitat Exclusion and Habitat Alteration

In addition to the direct impacts of suspended sediments on fish, increases in sediment loads can also alter fish habitat or the utilization of habitats by

fish (Scullion and Milner 1979, Lisle and Lewis 1992). High sediment loads can alter fish habitats temporarily by affecting water quality, making a stream reach unsuitable for use by fish. This habitat exclusion, if timed inappropriately, could have impacts on fish populations if the affected habitat is critical to the population during the period of elevated sediment load. This principle of habitat exclusion is a very important one; however, this issue is separate from the issue of direct habitat alteration that will be discussed below.

Sediment episodes can have a prolonged effect on the suitability of habitats within a stream reach through increased levels of sedimentation. In fact, sedimentation is the single most important effect associated with sediment load increases, because sediment loads can alter the gross morphology of streams as well as the composition of the stream bed and associated habitats.

Changes in Stream Bed Porosity

Larger-sized materials, such as fine to coarse sand are quick to settle onto the stream bed. This material may accumulate on the surface of the stream bed or filter down into the inter-gravel spaces. Interstitial spaces can become clogged by the downward or, to a lesser extent, by the horizontal movement of sediment (Beschta and Jackson 1979).

Water movement through the stream bed materials is important for the benthic communities that reside there, and for the developing embryos of fish species who bury their eggs. Inter-gravel water movement is controlled by several hydraulic and physical properties of the stream and its bed. The permeability of the stream bed is determined by size composition of the substrate material, viscosity of the water (temperature dependent) and the packing of the substrate material (Stuart 1953; Cooper 1965). A small increase in the proportion of fine material can severely reduce the porosity and permeability of the gravel bed (Lisle and Lewis 1992) and the ability of alevins to receive adequate oxygen and emerge from the gravel.

Changes in Stream Morphology

In addition to altering stream bed composition, elevated sediment loads can also change channel geometry (Klein 1984). Elevated levels of sediment deposition can reduce the depth of pools and produce a net reduction in riffle areas. This accumulation of streambed deposits can reduce available habitat. For example, deposition of sediments in pools and other areas of instream cover can cause a decrease in the fish holding capacity of a stream reach (Bjornn et al. 1977). Smith and Saunders (1958) found that decreased brook trout (*Salvelinus fontinalis*) populations were related to infilling of available cover. Alexander and Hansen (1988) also noted than a decrease in sand bedload sediment was associated with an increase in rainbow trout and brown trout (*Salmo trutta*) populations. Changes in physical morphology of the stream can also inhibit the movement of fish or change the distribution of adult fish (Alabaster and Lloyd 1982).

Channels affected by sediment derived from Anthropogenic disturbance are also more transitory in nature. Fox (1974) found urban watersheds exhibited a 33% monthly change in geometry as compared to a 5% change in less disturbed rural drainages. Sediment material deposited within streams can be in constant motion as bedload transport slowly moves the deposited materials through the system. This material in motion can increase bed scour and bank erosion as the sediment increases the erosive force of the water, by creating a sand blasting effect.

Sedimentation Effects on Spawning Habitats

River spawning salmonids typically deposit their eggs in gravel beds commonly found in the upper reaches of river systems. For example, brown trout typically bury eggs in interstitial spaces of the substrata to depths of 9 to 12 cm (Scullion and Miller 1979). Alevins remain in the interstitial spaces until the start of exogenous feeding. The percolation of water through the incubation substrate is an essential factor in determining the survival rate of incubating eggs (Lisle and Lewis 1992).

An increase in percent of fine material in the stream bed can have impacts on egg survival rates (Shaw and Maga 1943; Cordone and Kelley 1961) because it reduces streambed permeability. Lowered permeability reduces the interchange between stream flow and water movement through the redd, resulting in a reduction in the supply of dissolved oxygen to the egg and a hindrance to the removal of metabolites. Slaney et al. (1977) reported that rainbow trout egg survival was significantly reduced when spawning gravel contained more than 3% of fines (diameter 0.297 mm). In addition, Hall and Lantz (1969) determined that hatching success of coho salmon and cutthroat trout (*Onchorhynchus clarki*) was reduced by 40 to 80% when spawning substrates contain 20 to 50% fines (1–3 mm diameter).

Even if intergravel flow is adequate for embryo development, sand that plugs the interstitial areas near the surface of the stream bed can prevent

alevins from emerging from the gravel (Koski 1966; Phillips et al. 1975). For example, the emergence success of westslope cutthroat trout was reduced from 76 to 4% when fine sediment was added to redds (Weaver and Fraley 1993).

Female stream spawning salmonids typically clean an area of the stream bed in which they bury their eggs. This nest building activity flushes sediments and increases the stream bed permeability. With time, sediment conditions within the redd gradually return to ambient levels (Wickett 1954; McNeil and Ahnell 1964). Under normal conditions, this slow increase in sediment intrusion is not a problem; however, increased levels of sediment within a system as a result of anthropocentric disturbance increase the rate and level of sediment intrusion and reduces the period of time in which the redd is clean. The period of time before sediment intrusion into the redd is very important with respect to the survival of salmonid larvae. Studies by Wickett (1954) suggest that sediment accumulation during early embryonic development may result in higher egg mortalities than if deposition occurs after the circulatory system of developing larvae is functional. This may be due to the higher efficiency in oxygen uptake by the embryo or alevin with a functional circulatory system (Shaw and Maga 1942).

Ringler and Hall (1975) documented increased temperature and reduced dissolved oxygen levels of intragravel water in salmon and trout spawning beds because of clearcut logging practices. An associated reduction in resident cutthroat trout populations was attributed to this reduction in spawning habitat suitability. However, the failure to document serious reductions in coho salmon could be related to sediment clearing and removal by these larger fish during redd construction.

Sedimentation Effects on Fish Rearing Habitat

Sediment deposition also affects rearing habitat of juvenile fish because young salmonids frequently use the interstitial spaces in the stream bed for cover. Thus, a reduction in the suitability of potential rearing habitat as a result of sediment introduction is related to a reduction in the space available for occupancy (Reiser et al. 1985). When pools and interstitial spaces in gravel fill with sediment, the total amount of habitat available for rearing is reduced (Bjornn et al. 1977). Griffith and Smith (1993) found that numbers of juvenile rainbow trout and cutthroat trout decreased due to lack of available cover in heavily embedded gravel substrata. Interstitial space is particularly important during winter because juvenile fish live in these areas making them especially susceptible to impacts from increased sedimentation (Bjornn et al. 1977). Without these inter- gravel refugia, young fish may abandon the stream or move to less suitable areas where survival rates may be reduced.

Sedimentation Effects on Food Supply

Sedimentation can affect fish populations by altering the available food supply. Increased concentrations of suspended sediments and increased rates of sedimentation can reduce the primary productivity of the impacted area. Periphyton communities are likely the most susceptible to the scouring action of suspended particles or burial by sediments. At concentrations exceeding 115 mg/L, suspended sediments can reduce light penetration and primary productivity (Singleton 1985). A reduction in primary productivity has the potential to appreciably decrease the food supply of macrobenthos that graze on periphyton (Newcombe and MacDonald 1991). Many macrobenthic organisms are, in turn, used as a food source by fish.

Increased sediment loads in streams can also have an affect on zooplankton and macrobenthos. Sediment release can affect the density, diversity and structure of resident invertebrate communities (Gammon 1970; Lenat et al. 1981). A number of studies have demonstrated decreases in invertebrate densities and biomass following sedimentation events (Wagener 1984; Mende 1989). Increases in sediment input may reduce the density of invertebrates by directly affecting aspects of their physiology or by altering their habitat. Suspended sediments can have an abrasive effect on invertebrates and interfere with the respiratory and feeding activities of benthic animals (Tsui and McCart 1981). Increased sediment deposition may also reduce the biomass of invertebrates by filling the interstitial spaces with sediments and by increasing invertebrate drift or covering the benthic community in a blanket of silt (Cordone and Kelley 1961; Tsui and McCart 1981). Increases in sediment deposition that affect the growth, abundance, or species composition of the periphytic (attached) algal community will also have an effect on the macroinvertebrate grazers that feed predominantly on periphyton (Newcombe and MacDonald 1991).

A change in particle-size distribution in the stream bed can alter the habitat and make it unsuitable for certain species of invertebrates. Gammon (1970) noticed that an increase in suspended sediments from 40 to 120 mg/L resulted in a 25 to 60%

decrease in the density of stream macroinvertebrates. Likewise, Slaney et al. (1977) found that a 16-hour pulse of suspended sediments (2500 to 3000 mg/L) led to a 75% reduction of invertebrate biomass within the most affected areas.

Sedimentation can alter the structure of the benthic invertebrate community by causing a shift in the proportion from one functional group to another. For example, streams with clear water normally contain a high proportion of invertebrates in the shredder group; however, if sediment deposition is substantially increased, shifts to other groups such as grazers (Bode 1988) or collector-gatherers may occur (Wagener 1984). Some studies indicate that increased inputs of sediments cause a shift towards chironomid-dominant benthic communities (Rosenberg and Snow 1975; Dance 1978; Lenat et al. 1981).

Benthic fauna possess behavioral and morphological adaptations that limit them from being displaced in a unidirectional flow environment (Hynes 1973). Invertebrate drift, however, is a continuous redistribution mechanism that occurs in most stream ecosystems. It is an important factor in the regulation of population density (Williams and Hynes 1976), in the dispersion of aggregations of young larvae (Anderson and Lehmkuhl 1967), in the abandonment of unsuitable areas (Williams and Hynes 1976), and in the recolonization of areas after disturbance (Barton 1977).

Invertebrate drift may be induced by elevated suspended sediment levels (Rosenberg and Weins 1978). Increased rates of downstream drift by macrobenthos can be induced by concentrations as low as 23 mg/L (Rosenberg and Snow 1975). Drifting affords invertebrate taxa that are sensitive to increased sediment loads the opportunity to avoid areas that become unsuitable as a result of high suspended sediment levels. Conversely, invertebrate drift is considered to be the most important component of ecosystem recovery following stream disturbances (Williams and Hynes 1976; Barton 1977; Young 1986). This is especially true in areas of swift-flowing waters (Waters 1964).

Sedimentation Effects on Overwintering Habitat

The magnitude of impact upon fish resulting from increased concentrations of suspended sediments and levels of sedimentation can vary seasonally. It has been argued that the lowered metabolic requirements during winter conditions may in some ways provide a protective influence to conditions such as gill trauma and decreased gill function (C. Newcombe, BCMELP, personal communication). However, the ability of the fish to compensate for the stress of suspended sediments is influenced by a number of factors including the physiological condition of the fish and its ability to respond to the stress.

Early live stages (i.e., eggs, alevins) of many salmonids are found in the stream bed during the winter months. These stages are particularly sensitive to the effects of increased concentrations of suspended sediments and the deposition of fines sediments. The introduction of sediments during the winter, therefore, has the potential to appreciably influence these early life stages.

Bjornn et al. (1977) found that the number of juvenile salmon that a stream can support in winter was greatly reduced when the inter-cobble spaces were filled with fine sediment. The decreased carrying capacity was a function of both a loss of substrate cover for juvenile fish and a reduction in food as benthic invertebrate communities changed. Bjornn et al. (1977) suggested that the summer rearing or winter holding habitat may be more influential to the carrying capacity of a stream reach than embryo survival.

During winter fish generally experience decreased energy reserves and as such search for habitat that allows them to reduce energy expenditures (Clapp et al. 1990; Nickelson et al. 1992). Preferred habitats are species dependent; however, for most salmonids preferred habitats are located in low-velocity areas such as pools and behind instream cover were focal velocities are low (Vondracek and Longanecker 1993; Griffith and Smith 1993; Modde et al. 1991; Heggenes and Saltveit 1990; Cunjak and Power 1986; Tschaplinski and Hartman 1983). By remaining in low velocity areas, fish are able to minimize their energy expenditures and hence reduce the rate of metabolic depletion (Cunjak and Power 1986).

Land-use activities that increase the delivery of fine materials to streams can significantly affect the overwintering survival of resident fish. A mechanism of potential impact is a depletion of critical energy reserves as a result of increased physiological stress, alterations in behavior and/or exclusion from preferred sites of overwintering habitat. This is particularly deleterious to fish species and life stages that prefer to overwinter within the interstitial spaces of the stream bed. The net loss in energy reserves will depend on the concentration of sediment and the duration of impact. Dependent upon existing energy reserves, the fish may be able to tolerate the energy depletion attributable to an increase in the cough reflex and reduced feeding, but may not be able to tolerate the energy depletion associated

with displacement from critical habitats.

Preferred winter habitat areas of low current velocity are often predisposed to sedimentation (Cunjak 1996), and lower flows often experienced during winter may result in higher rates of sediment deposition. Bjornn et al. (1977) found, during sediment experiments, that the spring freshet from snow melt was rarely sufficient to transport sediment out of pools; therefore, the damage to these areas is frequently of a longer-term than sediments deposited in more erosive habitats. Due to natural factors, the availability of winter habitat is generally less than that of summer habitat and may be more influential in the determination of the stream's natural carrying capacity (Cunjak 1996; Mason 1976). A further reduction in the abundance of already limited winter habitat may significantly affect the overall fish population of a watercourse (Hartman and Scrivener 1990). Additive to this problem may be a reduction in food supply resulting from benthic drift or burying of food supplies. Elwood and Waters (1969) observed that increased sedimentation reduced the population of invertebrates and hence the capacity of the stream to support brook trout. A reduced food supply, and a greater expenditure of energy in food search and avoidance of higher concentrations of suspended sediments may significantly impact upon the fish's ability to compensate for negative physiological changes and the ability to survive the winter.

Sediment Load Biological Response Relationships

The Dose/Response Approach

One method that has been developed to address the issue of quantifying the adverse effects of TSS on fish is the ranked effects model first put forward by Newcombe and MacDonald (1991). This model compiled information from more than 70 studies on the effects of inorganic suspended sediments on freshwater fish (mainly salmonids) and invertebrates, and ranked the severity of impacts from 1 to 14 (rank effects). Linear regression was used to correlate ranked effects with intensity (concentration x duration) of increased suspended sediment load (Newcombe and MacDonald 1991). Because the effect of elevated TSS levels on fish is a function of both the concentration of suspended sediment and the duration of the exposure, Newcombe and MacDonald (1991) developed a Severity Index (SI). This index provides a standardized relative measure of exposure. It is the natural logarithm of the concentration ($mg{\cdot}L^{-1}$) multiplied by hours of exposure (i.e., Ln $mg{\cdot}h{\cdot}L^{-1}$). This SI provides a convenient tool for predicting effects of episodes of elevated suspended sediments of known concentration and duration.

The usefulness of the Newcombe and MacDonald (1991) concentration-duration response model has been questioned in the past (Gregory et al. 1993). The main concerns with the approach are the highly variable nature of the data used to develop the severity of effects model that reduces its predictive power, and the concern that the model is unrealistically simplistic (Gregory et al. 1993). A separate concern associated with the Newcombe and MacDonald (1991) severity of effects model is the severity index (Ln $mg{\cdot}h{\cdot}L^{-1}$) assumes a unit increase in episode duration (in hours) has a similar effect as a unit increase in concentration (in mg/L) (Anderson et al. 1996).

In 1994, the ranked effects model was further refined (Newcombe 1994). Using 140 articles on suspended sediment pollution, Newcombe developed a database of nearly 1200 datapoints concerning the effects of suspended sediments and associated effects upon marine and freshwater biota. With this database, regression analysis was used to relate severity of effect to the dose of TSS for specific fish species or assemblages. This approach was used to describe the dose/response relationship for the effects of suspended sediments on salmonid fishes (Newcombe and MacDonald 1991), on juvenile salmon (Newcombe 1994), and for other coldwater fishes (Newcombe 1994) using sub-sets of the dataset that are presented in complete form in Newcombe (1994).

In addition to the dose/response relationships presented for salmonid fishes (Newcombe and MacDonald 1991) and coldwater fishes and underyearling trout (Newcombe 1994), Newcombe and Jensen (1995) further expand upon dose/response relationships of aquatic resources to sediment exposure. Newcombe and Jensen (1995) presented six sediment dose/response relationships for specific fish communities exposed to elevated sediment loads. One of the important additions to the analysis presented in Newcombe and Jensen (1995) was the linkage of grain size of sediment to the nature of ill-effect associated with sediment exposure. The dose/response relationships presented in Newcombe and Jensen (1995) were organized according to four variables: taxonomic group; life stage; natural history and estimated predominant particle size range of the sediment episode. The dose/response relationships were characterized by three variables: [x], [y], and [z], where:

[x] = duration of exposure expressed as the natural log of hours;

[y] = concentration of sediment expressed as natural log of mg/L;

[z] = severity of ill effect (SE).

Since the work of Newcombe and MacDonald (1991) and Newcombe (1994) used the Stress Index (Ln concentration · duration), the dose/response relationships presented were of the form:

Equation 1: $z = a + bx$

where:

SE = severity of ill effects

a = intercept;

b = slope of the regression line

x = stress index value

The six dose/response relationships presented in Newcombe and Jensen (1995) do not characterize response using the stress index, and are presented in the form:

Equation 2: $z = a + b(\mathrm{Ln}\ x) + c(\mathrm{Ln}\ y)$

The dose/response relationships (Newcombe and MacDonald 1991; Newcombe 1994; Newcombe and Jensen 1995) provide insight into the relationship between sediment release and adverse effects on a variety of fish communities. These relationships provide increased precision for the prediction of response of a particular fish species or assemblage of species based on a given dose of suspended sediment.

The database used to determine these relationships relied heavily on the physiological response of fish to increases in sediment load; therefore, the relationships presented may not be directly applicable to the prediction of physical alteration to fish habitats due to sediment load increases and increased sedimentation nor the long-term impacts from reduced growth or the possible exclusion from certain habitats.

Developing the Habitat Effects Database

Anderson et al. (1996) attempted to develop effects relationships for fish habitat response to increased sediment. The first step in developing more specific dose/response criteria for habitat effects was to search the literature for studies reporting TSS concentrations and their effects on fish habitat. Several fundamental criteria had to be satisfied for data from a report to be included in the expanded database that was developed. The report had to give at least: 1) a concentration of TSS; 2) a duration of exposure to 3) one or more identified species of fish; and 4) a description of the effect. The severity of effect, and occasionally other data, sometimes had to be inferred from the qualitative descriptions. A total of 18 reports, containing some 53 new documentations of TSS effects, were retrieved in the literature search (Anderson et al. 1996).

Dose/Response Relationships of Freshwater Habitats to Sediment Releases

Severity of ill-effects rankings on a scale from 0 to 14 were assigned to each documented effect following the severity-of-ill-effects scale published by MacDonald and Newcombe (1993), Newcombe (1994), and Newcombe and Jensen (1995). The nature of the observed habitat effect was assigned one of the following class effect rankings:

SE = 3 Measured change in habitat preference;

SE = 7 Moderate habitat degradation, measured by a change in the invertebrate community;

SE = 10 Moderately severe habitat degradation, as defined by measurable reductions in the productivity of habitat for extended periods (months) or over a large area (kilometres);

SE = 12 Severe habitat degradation, as measured by long-term (years) alterations in the ability of existing habitats to support fish or invertebrates; or

SE = 14 Catastrophic or total destruction of habitat in the receiving environment.

Emphasis was placed on sediment release events (i.e., effects of sediment pollution events rather than chronic erosion and sediment load problems). The database excluded any datapoints for which the extent of habitat modification could not be ascertained from the primary manuscripts. The database was reduced to 35 entries (Anderson et al. 1996) and was used in the analysis described below to develop a relationship between sediment dose and habitat effects.

The dose/response approach of Newcombe and MacDonald (1991) and Newcombe (1994) defines dose as the product of TSS concentration (C in mg/L) and duration of exposure (T in hours). This definition of dose is strictly empirical and reflects the observation that the product of concentration and duration bears a closer correlation with ranked effects than concentration alone. The inherent assumption is that brief exposures to high doses of

TSS are equivalent in effect to prolonged exposure of much lower doses. Because severity of effect is determined based on a linear relationship with dose (Ln C·T) of the form SE = a + b (Ln C·T), the biological receptor response (SE) is assumed to respond to an effective dose in which concentration and duration are equally as important (i.e., the Effective Dose = C^nT; where n = 1).

However, it has been proven in much of the literature related to the response of biological receptors to toxic agents, that the relationship between concentration and duration is often more complex. That is, a high concentration for a very short time can cause a higher or lower response than can a low concentration for a longer time (Zelt 1995). In essence, by assuming a linear response (as measured by SE) to dose (as a function of Ln C·T), it is assumed that a unit increase in concentration (in mg/L) is equal to a unit increase in time (in hours). This assumption may or may not be a valid one. In an effort to address the potential for non-linearity in the relationship between concentration and duration in determining the effective dose of sediment (i.e., Effective Dose $=C^n \cdot T$; where $n \neq 1$), Newcombe and Jensen (1995) used multiple regression analysis to develop severity of effect relationships based on concentration and duration:

Equation 3: $z = a + b(\text{Ln } x) + c(\text{Ln } y)$

This approach, in effect, allows for different factors (slopes) to be assigned separately to the variables of concentration and duration.

In order to explore the relationship between concentration and duration in influencing habitat change, Anderson et al. (1996) also used multiple regression analysis to analyze the habitat effects database. This analysis identified a relationship between sediment exposure and habitat effects that can be described by the equation:

Equation 4: $z = 0.637 + 0.740\text{Ln}(X) + 0.864\text{Ln}(Y)$;

$r^2(\text{adj}) = 0.627$; n=35; $p<0.001$.

Statistics for the multiple regression relationship presented are summarized in Table 1. The "T" statistic for each slope in the regression (Table 2) is an expression of the importance of each variable with respect to the relationship derived. The higher score attributed to duration indicates its importance in determination of habitat effects. This indicates that concentration and duration affect the extent of habitat alteration in dissimilar ways or in other words, that the effective dose of sediment is a function of a non-linear relationship between the two predictive variables (i.e., Effective Dose $=C^n \cdot T$; where $n \neq 1$).

Discussion

Confounding Factors

The dose/response relationships that have been developed make generalizations about the anticipated level of effects to the aquatic environment that may result from elevated sediment levels. Because these are generalizations, the actual effects that are realized by a sediment release episode may be more or less severe based on a number of confounding factors.

The potential for adverse effects on fish and their habitats associated with sediment release is a function of increasing particle size (Newcombe 1996). More information relating dose-response relationships between specific fish guilds or habitat types as a function of particle size range is required in order to develop a better understanding of this confounding factor.

The angularity or mineralogy of suspended particles may play an important role in the potential for physiological or toxicity effects (Newcombe and Jensen 1995). The angularity of a particle may be of particular importance with respect to gill abrasion of fish within the receiving environment, and may also influence the rate of infiltration of particles into the stream bed. Meanwhile, the mineralogy of the particle may be important because the particle itself may have some potential chemical activity at the cellular level (Newcombe and Jensen 1995). In addition, the potential for contaminants adsorbed to sediment particles is also a concern, because contaminated sediments could have more dramatic effects than those that might be caused by the increase of sediment load alone.

Table 1. Statistics for the Multiple Regression Relationship (Equation 4)

Variable	Coefficient	Standard Error	Standard Coefficient	Tolerance	T	P (2 tailed)
Constant	0.0637	1.293	0.000	–	0.493	0.625
Ln Con.	0.864	0.176	0.520	0.973	4.903	0.000
Ln Duration	0.740	0.111	0.706	0.973	6.652	0.000

The amount of material that intrudes into the gravel bed has been shown to be highly dependent on the grain-size distribution of the transported sediment as well as that of the gravel bed. If the suspended sediment load is composed of very fine material, the gravel pores tend to fill from the bottom to the top of the pavement layer. If the suspended particles are larger in size, angular or platelet in shape, a film can develop within the substrata, which will tend to limit the intrusion of additional sediments into the interstitial spaces of the stream bed (Beschta and Jackson 1979). Beschta and Jackson (1979) concluded that the finer the suspended sediment, the greater the potential was to fill interstitial voids.

The shape of the stream bed substrata may also affect sediment deposition. Under low flow conditions, rounded stream bed substrata tend to accumulate more sediment than angular substrata, whereas, during high flows, the reverse is true (Meehan and Swanston 1977). This may be due to the reduced turbulence levels at the gravel bed in rounded stream beds during low flows, while at higher discharges a flow separation zone can develop behind angular materials causing greater sediment deposition (Reiser et al. 1985).

The temperature of the water can have an impact on the severity of the effects caused by a sediment release event. The oxygen holding capacity of water and the metabolic and respiratory rates of fish are influenced by water temperature. Consequently, the effects of sediment exposure may be greater in seasonably warm waters than in seasonably cold water (Newcombe and Jensen 1995). However, during winter conditions aquatic organisms may be especially vulnerable to additional stressors. Since stress has been defined as the sum of all physiological responses, the severity of effects that are caused by sediment release will be a function of the level of stress at the time of, or before, the period of elevated sediment load.

Factors Influencing the Risk of Habitat Alteration

Many attributes may influence the extent of sediment-induced aquatic habitat alteration. The principal factor that influences the extent of habitat alteration is the increase in sediment load associated with watershed disturbance. In addition, the sensitivity of the exposed habitats and the length of time habitats are likely to be impaired are also important factors that influence the level of habitat alteration.

Considerations regarding the sensitivity of the receiving environment include the susceptibility to alteration of the habitats within the receiving environment, and the timing of the elevated sediment loads. In addition to overall sensitivity of the watercourse and the sensitivity of the habitats it supports, a related consideration is the species and life stages that may be present during times of instream activity associated with forest harvesting or road construction. Certain life stages are especially sensitive to increases in sediment load (such as developing eggs and larvae, or overwintering fish—particularly juvenile salmonid in streams); as a result, the presence of these life stages during instream construction would increase the sensitivity of the watercourse to disturbance.

An additional related consideration regarding construction timing sensitivity is the flow conditions within the watercourse during periods of elevated sediment loads. Watercourse discharge influences the concentration of suspended sediments, transport and deposition of materials as well as the extent of habitat present and the ability of resident biota to avoid areas of elevated sediment.

The duration of habitat impairment is one of the most critical considerations in relating the extent of habitat alteration to aquatic community impact because habitat alteration will only affect the aquatic community if the altered habitat would have been used during the period of impairment. Therefore, the level of concern associated with habitat alteration increases as the duration of impairment increases. The duration of impairment is considered to be the length of time before deposited sediments are flushed from the watercourse into a less sensitive area such as a lake and are normally viewed as the number of life history stages that are affected during the period in which the habitat is in an altered state.

As a result, evaluation of the extent of habitat alteration associated with elevated sediment loads must consider the level of sediment load increase (concentration and duration), the nature of the habitats and communities affected and the duration of likely impairment caused. The severity of effects approach, which have been developed by Newcombe and others, are not easily applied to the prediction of habitat change and attributing justifiable numbers to abstract concepts such as system sensitivity is difficult at best. Resource managers need to apply expert judgement to ensure that models and assumptions are not applied blindly and that model results do not violate the most important management tool, which is common sense.

Table 2. Logging road mitigation measures(taken from Waters 1995)

Design feature	Method	Purpose
Road placement	Avoid streams and steep slopes	Reduce erosion & mass soil movement
Road length	Few, short roads	Reduce total area of exposed roadbed
Road width	Narrow as practicable	Reduce area of disturbance
Road grade	5–15%, not flat, minimum 3% for drainage	Avoid rapid run-off
Road surface	Gravel, crushed rock roadbed	Reduce roadbed erosion
Cut slopes	Vertical or near vertical cut	Reduce excavation and erosion of slope
Fill slopes	Avoid road drainage and woody debris in fill	Stabilize fill slopes
Road drainage	Outslope drainage on shallow slopes, inside drainage on steep grades	Disperse drainage
Inside drainage	Ditch inside road	Carry run-off along road
Cross-drainage culvert	Underground pipe or log construction	Drain inside ditches or waterways
Water bar	Low hump, 30 ° angle downslope	Disperse run-off from roadbed
Broadbased dip	Wide drainage dip or bench	Disperse run-off from roadbed
Stream crossing	Minimize number, use appropriate method	Reduce direct aquatic impact
Vegetation planting	Seed grass, plant trees	Reduce exposed surface
Daylighting	Cut canopy to permit sunlight penetration	Promote drying of roadbed
Abandonment	Close access, remove crossings, install dips and water bars	Avoid subsequent use and maintenance

Mitigation Measures

Although most disturbances within a watershed inevitably increase the amount of erosion, the delivery of sediments to streams resulting from disturbances can be largely circumvented by proper design and planning. As discussed previously, logging road construction and operation can dramatically increase sediment loads in streams. The amount of disturbance caused by road construction and maintenance depends upon its design standard, gradient, total distance of road and intensity of use. For example, Megahan and Kidd (1972) indicated that proper siting and construction of roads could eliminate much of the sediment loading associated with mass erosion in steep terrain.

The density of logging road distribution can be a major factor in determining the associated increase in sediment loads in streams (Waters 1995). Cederholm et al. (1981) documented the greatest accumulation of fine sediments in streambeds associated with road areas that exceeded 2.5% of the total basin area. In addition, the length of logging roads also influences sediment delivery to streams. Cederholm et al. (1981) calculated total road lengths of 2.5 km of road per km^2 of watershed basin produced sediment more than four times natural rates. As a result, sediment mitigation measures have concentrated on the minimizing sediment associated with logging roads.

The logging system used is a critical factor in the determination of road density. For example, the use of jammer logging systems in the past has required a dense network of roads because these logging systems require a maximum road spacing of about 150 m. In areas of steep terrain, this approach to logging may disturb soil on 25% of the total logging area. High lead logging is a method that reduces the level of disturbance since a reduced road network is required to support this method. In steep topography areas in Idaho, high lead logging has reduced disturbance to less than 10% of the logging area. On the same types of slopes, jammer logging roads spaced 60–120 m apart disturbed 25–30% of the total area (Rice et al. 1974). Skyline logging systems permit an even wider road spacing of 500 m or more, depending on local conditions and topography. This reduces the area disturbed by 75% and may provide a greater buffer area for sediment filtration between the road and the stream channel (Megahan and Kidd 1972).

Skyline, balloon and helicopter systems have been developed to permit the logging of steep topography with a minimum amount of road disturbances (Rice et al. 1974). Binkley (1965) estimated that skyline yarding would save from 2.2 to 2.9 km of road over that required for high lead in a 1600-ha drainage area to be logged.

Control measures to minimize sediment delivery to streams associated with accelerated erosion within watersheds disturbed by forest harvest operations are of paramount importance in minimizing elevated sediment loads in streams. Because the best mitigation measure to minimize sediment loads in streams is to minimize the amount of erosion in the watershed, design features for access planning can be implemented that will minimize erosion associated with logging roads. Waters (1995) summarizes design features for the reduction of erosion from logging roads. This summary table is reproduced as Table 2.

Disturbance within watersheds inevitably increases sediment loads within adjacent watercourses. These increases in sediment load can have a substantial effect on fish and on their habitat. Through the proper planning and design of forest harvest systems, the level of sediments delivered to streams can be minimized allowing for the existence of both forestry and fish.

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The habitat database and habitat effects relationship presented in this paper were first developed and reported in Anderson et al. 1996. For more information on the database and the development of the relationship, the reader is directed to that manuscript. The author acknowledges the work of Waters (1995), and the on-going work of C. Newcombe. The author thanks the reviewers of this paper for their time and insightful comments and to staff of Golder Associates for their assistance.

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Application of a Coarse Woody Debris Recruitment Model to Investigate the Impacts of Riparian Forest Harvest and Stream Cleaning

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Abstract

Coarse woody debris (CWD), while often the primary source of structure in small streams, has been frequently subjected to intensive removal efforts with uncertain impacts. We present the integration of a growth and yield model, Forest Vegetation Simulator, with a mechanistic post-processor (CWD) to conduct long-term assessment of riparian forest harvest and in-stream CWD removal. The CWD model predicted a gradual recovery of CWD loads under some harvest scenarios 50–110 years after stream cleaning, but both selective harvesting and clearcutting decreased the volume of material entering the stream. The combination of stream cleaning and clearcutting generally decreased the in-stream CWD volume to < 10% of undisturbed levels. Depending on target thresholds for stream CWD loads, selective harvesting of riparian forests may deliver enough material to permit some manipulation.

Introduction

Over the last few decades, riparian zone management has experienced a renaissance in attitudes. Interest in preserving and facilitating natural structures has moved to the forefront of riparian planning as attitudes about aquatic ecosystem process have evolved. For example, we previously extracted great quantities of coarse woody debris (CWD, pieces of wood at least 10 cm in diameter and 1 m long) from rivers and streams to facilitate navigation and fish migration, often with unanticipated consequences (Swanson and Lienkaemper 1978; Harmon et al. 1986; Maser and Sedell 1994). However, more recent research has documented the importance of CWD in shaping stream habitat, providing cover for fish and invertebrates, organic carbon production, turbulent mixing of water, and channel morphology

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(e.g., Harmon et al. 1986; Bisson et al. 1987; Maser and Sedell 1994).

With increasing demand for forest products, managers are often faced with the dilemma of implementing forest management practices within sensitive riparian zones, yet frequently lack the tools to develop prescriptions that incorporate both timber and riparian interests. To alleviate some of these problems, we have developed a post-processor (CWD, v1.1) driven by a growth and yield model [the Forest Vegetation Simulator (FVS), Teton Variant, v6.1 (see Wykoff et al. (1982) for a general FVS description)] to predict riparian CWD recruitment through a combination of stochastic and deterministic processes. Originally calibrated for Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*)-dominated riparian forests on the Bridger-Teton National Forest in northwestern Wyoming, U.S.A., the CWD model is designed to assist managers interested in sustaining or recovering riparian CWD.

Methods

Study area description

During the summer of 1995, thirteen reaches were sampled on the Bridger-Teton National Forest to provide both model parameterization and stands for simulation. Small (first- to third-order) streams were selected because of the importance of CWD in these systems (Swanson et al. 1976), and because locally they had the highest likelihood of adjacent, reasonably continuous stands of Engelmann spruce and subalpine fir. The study streams (Table 1) were generally of low to moderate gradient (measured by change in elevation along the 250-m test reach) and low in sinuosity. In this part of the central Rockies, spring snowmelt floods and occasional summer thunderstorms dictate extremes in stream flow, which strongly influence in-stream CWD movement (Harmon et al. 1986).

Model development

The study reaches were sampled for a number of characteristics used to help parameterize the production and delivery mechanisms of CWD. Because a detailed explanation of how CWD works interactively with FVS is beyond the scope of this paper, we will only briefly summarize the process. The FVS generated the riparian stand dynamics required to run the post-processor. After stand data were entered in the appropriate input file format, FVS determined reproduction, growth, mortality, and harvesting through various procedures (*see* Wykoff et al. 1982; Ferguson and Crookston 1991; Crookston 1990). The FVS also produced an individual tree file[1] from which CWD extracted the necessary mortality information and snag characteristics. The CWD then determined (in this order) snag residency, snag failure, direction of snag fall, CWD fragmentation, and recruitment to the stream. Once delivered to the channel, within-stream CWD loads were derived from:

CWD volume = previous volume + CWD recruited - losses of CWD

Stream-borne losses of CWD typically arise from two sources: decomposition and transport, with high flows dominating CWD attenuation in many small streams (Harmon et al. 1986). For simplicity, we assumed that the negative flux of CWD (transport off-site + decay) was approximately equivalent to inputs from adjacent forests and deposition of material transported from upstream (we refer to this as the cyclic CWD turnover rate, *see* Table 2). This assumption yielded a steady-state CWD load in

Table 1. Properties of selected stream reaches on the Bridger-Teton National Forest, Wyoming, sampled during the summer of 1995. These streams are used in this paper for coarse woody debris production and delivery simulations.

Stream	Order	Drainage basin size (ha)	Mean bankfull width (m)	Reach sinuosity	Reach gradient (%)	Riparian forest density ($m^2 ha^{-1}$)
Moose Gulch Creek	1	414	3.4	1.12	2.0	23
Dry Lake Creek	2	1033	5.5	1.05	3.5	33
Hoback River	3	9883	14.5	1.02	1.5	31

[1] The FVS operates on a 10-year cycle; therefore, it generated growth and yield data on a cyclic (rather than annual) basis.

Table 2. Coarse woody debris (CWD) parameters for the stream reaches simulated for this study. Note that for undisturbed old-growth (this example), we assumed average cyclic CWD delivery equaled average cyclic CWD loss (steady-state conditions).

Stream	Mean bankfull width (m)	Current stream CWD load (m^3 100 m^{-1})	Average cyclic CWD delivery (m^3 100 m^{-1})	Cyclic CWD turnover rate[a]	Average cyclic CWD loss (m^3 100 m^{-1})
Moose Gulch Creek	3.4	21.1	2.7	0.13	2.7
Dry Lake Creek	5.5	8.6	2.3	0.26	2.3
Hoback River	14.5	11.9	3.2	0.27	3.2

[a] Cyclic CWD turnover rate is the ratio of average cyclic CWD loss to stream CWD volume at $t = 0$.

undisturbed streams, which, while unproven, is generally accepted for riparian CWD dynamics [*see* Froehlich (1973); Swanson and Lienkaemper (1978); Likens and Bilby (1982)].

Harvest simulation

The FVS is capable of simulating most traditional harvest practices repeatedly across broad time periods (up to 400 years), thus making it possible to evaluate the long-term consequences of management and estimate riparian CWD recovery for streams influenced by timber harvests and/or CWD removals. For this exercise, six different treatments were tested on a first-, second-, and third- order stream (Table 3). In low-intensity stream treatments (i.e., no in-channel debris removal), we assumed the streams transported a fixed percentage of CWD during each cycle regardless of CWD volume delivered. For higher intensity treatments (debris cleaning), we eliminated all CWD in the stream at the first treatment (t = 50 years), but not during subsequent harvests, thereby paralleling historical conditions when streams were previously cleaned (but not in conjunction with current or future harvests). We also assumed no slash was deposited into the stream as a result of harvesting.

The CWD model was run for 30 replicates (i.e., the FVS treelist output was processed 30 times for each scenario by CWD, and then averaged to generate the results in Figures 1–3) over a 300-year simulation period. Replication was needed to generate reliable CWD recruitment values because of the stochastic nature of some CWD subroutines, including snag location, direction of snag fall, and snag fragmentation. Each subject stand was modeled as 15-m strips of forest adjacent to the channel. Results are reported as CWD delivered in m^3 per 100 m of stream reach.

Table 3. Permutations of riparian forest harvests and stream coarse woody debris cleaning treatments used in this study (each was applied to all 3 stands)

TMT #1	No harvest, no stream cleaning
TMT #2	No harvest, stream cleaning
TMT #3	Clearcut, no stream cleaning
TMT #4	Clearcut, stream cleaning
TMT #5	Selective harvest, no stream cleaning
TMT #6	Selective harvest, stream cleaning

For stands that were cut, a target stocking[2] of 195 m^3 ha^{-1} was established as the threshold for harvest scheduling in FVS. We then evaluated the ecological and silvicultural effects of each treatment through differences in riparian CWD recruitment, net change in stream CWD loads, and differences in merchantable harvest volume.

Results and discussion

Predicted changes in stream CWD loads are presented in Figures 1–3. The no harvest, no streaming cleaning treatment (TMT #1) weakly oscillated over the length of the simulation, but generally followed the original (from year 0 to 50) steady-state CWD loads. While we lacked long-term field data on stream CWD, the simulated patterns of CWD loading generally followed natural fluctuations in stand density [barring catastrophic disturbance, see Bormann and Likens (1979) and Likens and Bilby (1982)]. Riparian CWD patterns changed drastically with stream cleaning, even under a no-harvest system. Complete stream CWD removal under unmodified canopies (TMT #2) resulted in decades-long lags in CWD volumes for all simulated streams (Figs.

[2] The target stocking of 195 m^3 ha^{-1} was used as a harvest threshold because it produced FVS-simulated rotation ages of 110–140 years, following silvicultural practices recommended by Long (1994).

1–3). Depending on initial steady-state CWD loads, stand productivity, and cyclic CWD turnover rates (Table 2), the CWD volume lag[3] for undisturbed riparian forests ranged from about 50 years at Dry Lake Creek to 80 years for the Hoback River to 110 years for Moose Gulch Creek. With only one stream CWD cleaning, these riparian forests eventually recovered TMT #1 stream CWD volumes. However, while their volumes may appear similar, it is unlikely that cleaned and uncleaned streams had functionally equivalent conditions during this relatively short simulation period. The dimensionality of CWD stored under TMT #1 is probably different from TMT #2, as the very large trunks and root masses present in TMT #1 are slow to accumulate and may persist for centuries in undisturbed systems (Swanson et al. 1976, Bisson et al. 1987). The post-treatment riparian CWD loads for TMT #2 are much more likely to be smaller pieces (on average) with different dynamics and structural properties.

The application of traditional harvest systems in riparian forests, even without stream cleaning, significantly depressed CWD recruitment and, inevitably, total riparian CWD volume. Selective harvest, here simulated by periodic thinnings of the largest size classes when stand volume exceeded 195 m^3 ha^{-1}, instituted a gradual but noteworthy decline in CWD volume as both mortality rates and the frequency of large trees declined. Increased tree vigor, generally thought to be a favorable aspect of managed forests, limited the potential amount of CWD recruited to the stream because weak trees were the most likely candidates to die under our assumptions. The removal of most of the largest live trees also decreased the volume of CWD entering the stream because these individuals were eliminated from the potential recruitment pool. Although not simulated by CWD, the change in piece dimensions may further reduce CWD storage as large logs play a critical role in retaining smaller, more transient pieces (Bisson et al. 1987).

Selection systems, however, yielded intermediate post-harvest CWD delivery because they reserved many residual trees > 10 cm d.b.h. (whereas clearcutting removed all CWD-potential stems and most advanced regeneration). Both harvest treatments that did not involve stream cleaning (TMT #3 and TMT #5) led to significant depletion (but not elimination) of riparian CWD. Retained riparian CWD likely included some large, stable pieces, providing the nucleus for post-harvest storage of smaller pieces. Harvest treatments with stream cleaning (TMT #4 and TMT #6) also experienced the extended riparian CWD storage lags noted under the no harvest, stream cleaning scenario (TMT #2). Our results further predicted that stream cleaning, when coupled with repeated clearcutting, virtually eliminated all CWD for decades and only recovered to minimal levels afterwards. Under these conditions, our analysis suggests that maintaining 10% of the riparian CWD volume of untreated streams (i.e., TMT #1) would be difficult (Figures 1–3).

While both harvest types significantly reduced CWD production from undisturbed conditions, projections indicated that selection systems can yield 68–104% of the clearcut timber volume of these same stands while retaining 38–65% of the CWD delivered to the streams (Table 4). Although we are aware of few studies relating efficiency or relative importance of specific riparian CWD loads for their functional value [although see Murphy et al. (1986) for a probable example], this level of recruitment may be sufficient to sustain relatively healthy levels of in-stream structure. While most federal and state agencies have regulations against engaging in high impact riparian treatments like CWD removal or clearcutting, private individuals or companies are not quite as fettered and may have an interest (especially in remote headwaters areas) of performing such treatments. It is also possible that governmental agencies may permit riparian zone harvests under limited situations (e.g., salvage harvests, wildlife improvements, remediation work), and therefore need to evaluate the impacts of treatment on CWD production and delivery. Perhaps the greatest potential of this post-processor lies with restoration of riparian CWD on systems that were severely harvested and cleaned decades ago and have yet to recover (see later recommendations).

Conclusions and recommendations

While the CWD model involves many assumptions and simplifies the processes involved in riparian CWD recruitment, it suggested trends likely to occur under the various treatments. Consideration of the CWD dynamics involved in these situations can be used to accommodate both timber and CWD production, as well as to repair prior damage. Vital to such efforts, however, is recognition of the uniqueness of each system and potential differences in response.

[3] Note that the post-stream cleaning lag represents the return to about 90% of the assumed steady-state CWD load at that point in time-achievement of identical loads requires an additional 50–100 years

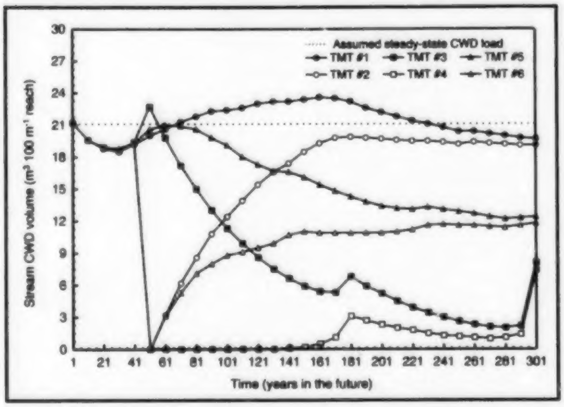

Figure 1. Stream coarse woody debris (CWD) load during the simulation, Moose Gulch Creek, Wyoming.

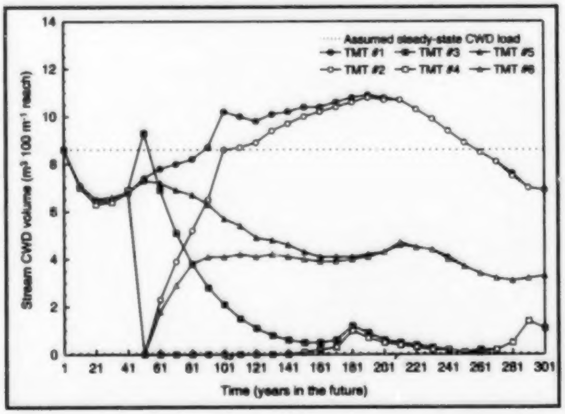

Figure 2. Stream coarse woody debris (CWD) load during the simulation, Dry Lake Creek, Wyoming.

With increasing demands on public forests, it becomes more critical to efficiently use available resources. In the Intermountain West, many of the most productive spruce–fir forests occur in riparian areas, which also serve as essential habitat for riparian-dependent species. This places additional pressure on managers, who must strive to meet often competitive goals with limited resources. We feel obliged to provide a few recommendations to hopefully reconcile some of the needs of timber management with riparian zone planning.

First and foremost, stream cleaning to improve stream flow is largely unnecessary and mostly deleterious. With the possible exception of logging slash added during harvest, channel CWD plays a significant positive role in sediment retention and habitat

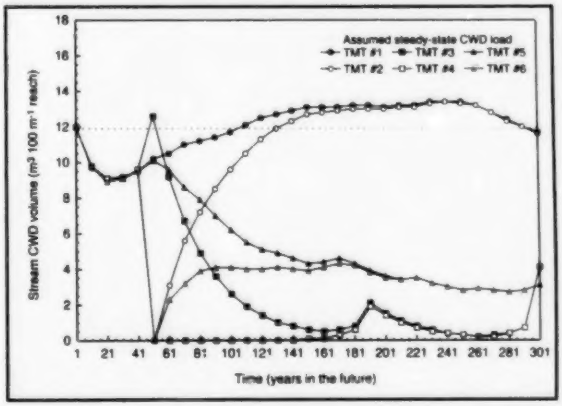

Figure 3. Stream coarse woody debris (CWD) load during the simulation, Hoback River, Wyoming.

Table 4. Forest Vegetation Simulator-predicted merchantable wood volume production versus cumulative (300 year) riparian coarse woody debris (CWD) recruitment for simulated treatments

Stream	Harvest treatment	Number of harvests	Cumulative CWD recruitment (m^3 100 m^{-1})	Relative recruitment (% of no harvest)	Merchantable harvest volume (m^3 ha^{-1})	Harvest total (% of clearcut)
Moose Gulch Creek	clearcut	3	24.4	29	264	100
	selection	4	54.8	65	180	68
	none	—	84.4	100	0	0
Dry Lake Creek	clearcut	3	12.5	18	254	100
	selection	5	34.8	51	215	85
	none	—	68.1	100	0	0
Hoback River	clearcut	3	20.5	21	373	100
	selection	8	37.1	38	389	104
	none	—	96.6	100	0	0

formation for many species, and rarely proves an obstacle for organisms. Stream cleaning has largely disappeared from land management practices in recent years, but could prove tempting to some operations in old-growth forests who view large CWD as a salvageable resource. We strongly recommend the recognition and retention of all natural CWD accumulations if riparian zones are to be harvested.

Clearcutting is probably also unadvisable in sensitive systems that are dependent on high CWD loads to provide structure or retain sediment. The primary impact of clearcutting on CWD recruitment (barring significant in-stream deposition of logging slash) is that it keeps adjacent riparian forests in early, vigorous successional stages, thereby minimizing the delivery of large, stable materials vital to healthy CWD stocking. Generally, selective harvests do not limit riparian forests to small diameter even-aged stands, and therefore may retain enough large trees to support substantial CWD delivery.

Managers may also desire a more aggressive approach to riparian CWD management. Retention of large, declining trees or girdling bigger, healthier trees in the immediate proximity of the stream, especially those whose lean favors entrance into the stream, should serve to supply large CWD during periods of post-harvest stand regrowth. Of course, each riparian system should be evaluated on a case-by-case basis, but riparian forest management and preservation of many stream structural qualities need not be mutually exclusive events.

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Use of Radio Telemetry to Identify Movements, Habitat Use, and Spawning of Brook Trout in the Copper Lake Watershed, Newfoundland, Canada

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Abstract

Movements of 19 brook trout (*Salvelinus fontinalis*) in two lakes (17.5 and 82.4 ha) within the Copper Lake watershed, Newfoundland were monitored between August 10 and October 7, 1995 using surgically implanted radio transmitters. Movement of fish was variable, with 88% exhibiting ranges less than one-third the area of their 'home lake'. Three fish moved into tributary streams to spawn and none moved between lakes. Examination of telemetered locations and movements during the spawning period (late September to early October) indicated an association with potential lacustrine spawning sites. This was subsequently confirmed by the observation of a large number of redds at these locations. Lake spawning of brook trout in Newfoundland has rarely been documented, and these observations will assist in evaluating the relative importance of lacustrine and fluvial spawning habitats.

McCarthy, J.H., Scruton, D.A., Green, J.M., and McKinley, R.S. 1998. Use of radio telemetry to identify movements, habitat use, and spawning of brook trout in the Copper Lake watershed, Newfoundland, Canada. Pages 515-521 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest- fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

The effects of forest harvesting practices on fluvial habitat poses serious implications for the management of fisheries and forests in Newfoundland (McCarthy et al. 1998). The federal department of Fisheries and Oceans Canada (DFO) fish habitat management policy outlines a 'no net loss' philosophy in maintaining the productive capacity of fish habitats (Fish Habitat Management Branch 1986). Integral to this is the maintenance of spawning habitat.

Meehan (1991) reviewed many facets of salmonid spawning habitat that can be adversely affected by forest harvesting activities. Major factors include changes in i) substrate composition (sedimentation); ii) suspended sediment; iii) hydrological regimes; and iv) temperature profiles.

Brook trout (*Salvelinus fontinalis*) are considered classic fluvial spawners (Scott and Crossman 1979); however, shoal or lake spawning has been described (Witzel and MacCrimmon 1983; Fraser 1985; Chapman 1988; Schofield 1993). Lacustrine spawning of brook trout has rarely been documented in Newfoundland (Cowan and Baggs 1988), and hence, its importance to the reproductive capacity of populations is unknown. With this in mind, 19 resident brook trout were implanted with transmitters to determine lake-use and spawning habitat of larger fish in the Copper Lake watershed. Movements of these individuals may help determine not only where suitable spawning habitat is located but also the behavior patterns associated with redd site selection (i.e., if trout actively search large areas for suitable sites or if they quickly approach and remain in areas with suitable spawning habitat once spawning begins).

Materials and Methods

Study Site

The Copper Lake watershed is located about 17 km southeast of Corner Brook, Newfoundland, Canada (48°49'17.5"N, 57°46'27.0"W) (Fig. 1). The forest in this area has not been previously harvested (Scruton et al. 1995). This watershed was chosen for a large buffer zone research project that will eventually evaluate the effectiveness of a 20-m no-harvest buffer zone for the preservation of water quality and aquatic ecosystems. The two lakes available to fish

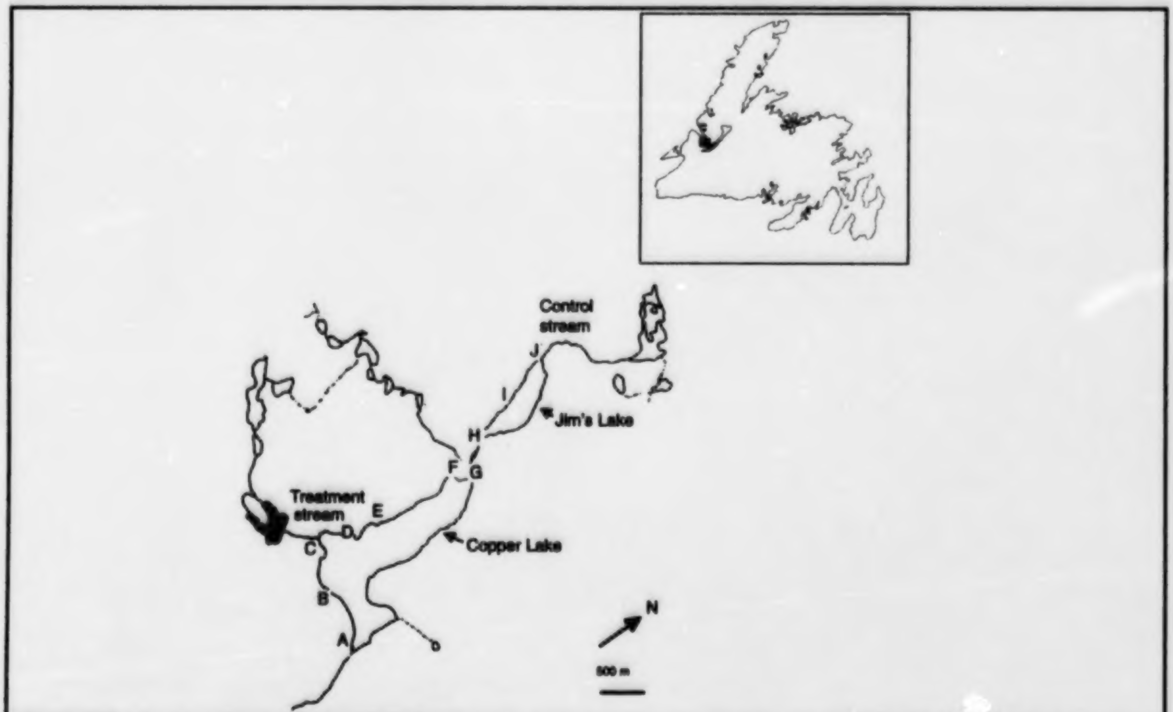

Figure 1. The location of the Copper Lake watershed, tributary streams, clear-cut (shaded), and fixed telemetry locations (letters) within the watershed.

Table 1. Estimated distance (m) from the previous position for each fish tracked in Copper Lake, Newfoundland

Date	Transmitter number								
Month/day	045	085	105	126	165	225	245	285	306
8/11									
8/15	25	25							
8/16	150	400	550	500					
8/17	0	650	1000	500					
8/18	100	250	0	200	1000				
8/19	350	300	550	300	100				
8/21	0	0	250	700	75				
8/22	0	150	200	*[a]	0				
8/24	25	150	25	550	850	0	800	50	800
8/28	25	250	150	100	400	*	750	350	750
8/29	25	250	100	150	500	*	25	250	D[b]
9/03	25	50	100	100	100	*	*	400	
9/04	0	0	0	200	*	*	0	*	
9/05	0	500	200	400	0	*	100	100	
9/07	0	25	100	300	800	*	*	0	
9/11	0	400	200	500	500	*	25	100	
9/12	0	0	1250	200	150	*	800	500	
9/13	0	500	700	700	1250	*	50	600	
9/19	0	1100	*	*	1000	*	500	800	
9/20	0	1000	400	1300	450	*	0	100	
9/21	0	1000	*	500	1300	*	300	0	
9/25	150	700	400	0	*	*	700	600	
9/26	0	400	400	200	800	*	200	200	
10/02	75	450	300	900	350	*	200	300	
10/04	75	1000	300	350	200	*	300	0	
10/05	0	0	600	500	800	*	200	200	
10/06	0	100	0	550	*	*	0	0	
10/07	100	250	500	0	250	*	300	400	

[a] * = No signal detected.

[b] D = Fish was found dead.

within the study area were Copper (82.4 ha) and Jim's (17.5 ha) lakes (Fig. 1). All headwater ponds were inaccessible to trout from the two lakes due to obstructions (waterfalls). In 1993, the watershed was surveyed and a cutting regime consistent with the experimental study design of the research project was designed (Scruton et al. 1995).

Surgical Implantation

Fish were sampled from Copper and Jim's lakes with fyke nets and by angling (barbless hooks). Radio transmitters (Lotek model no. FSM-3) had an expected battery life of 60 days and weighed 2.3 g in water. All transmitters were implanted between August 9 and 24, 1995. They were surgically implanted using the method described by McKinley et al. (1992) with the following exception: the incision for the transmitter was made on the ventral surface immediately posterior to the pelvic fins and anterior to the anus. This area provided more muscle for suturing and had less tendency to tear after the sutures were in place. After surgery, fish were allowed to recover in small impoundments within the lakes for 0.25–0.5 h before being released back into the lake from where they were captured. Fish were tracked from August 10 to October 7, 1995. In total, 10 fish from Jim's Lake and nine fish from Copper Lake were studied.

Fish were tracked with a hand-held receiver (Lotek model no. SRX-400) and a Yagii antenna from fixed-land positions on a daily basis (between 1300 and 2000 h) (Fig. 1). Subsequent triangulation gave the point-location of each fish and an estimate of the minimum distance traveled since the last known position. The minimum distance was the linear distance between the present and last known point-location. The point-locations were plotted on topographical maps of the watershed to determine habitat use

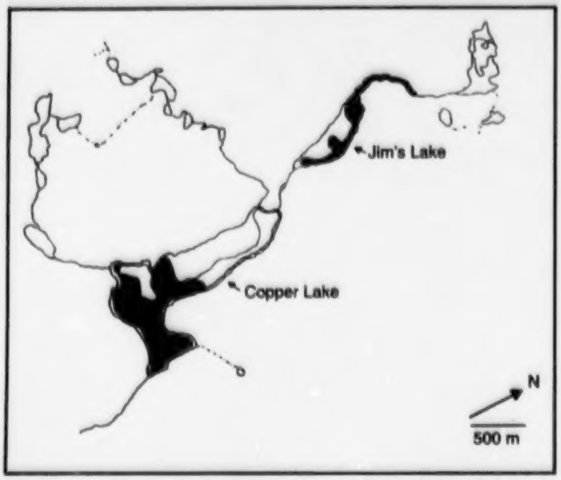

Figure 2. Representative ranges of implanted brook trout.

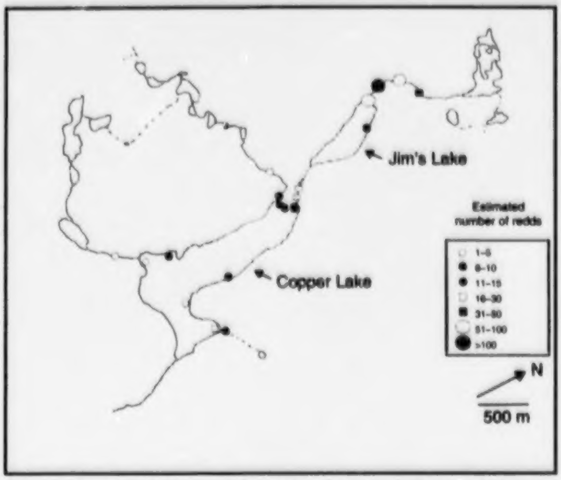

Figure 3. Location and estimated number of lacustrine and fluvial redds within the study area.

and range of movements for the duration of tracking. Individual ranges were grouped into representative ranges, which were mapped for each lake.

Spawning Activity

Spawning activity within the watershed was monitored from September 27 to October 7, 1995, by surveying the watershed for redds. All streams were monitored by walking along stream banks while the lakes were surveyed by boat. All redds within the streams were counted. However, the maneuvering of the boat and varying depth of water in the lakes made an actual count of the redds difficult. Therefore, a visual estimate of the number of redds present in each area was determined as best as conditions would allow. These were later mapped. Redds were identified as light patches of substrate that had been cleaned of the surface covering of filamentous green algae and debris (Cowan and Baggs 1988). These were later related to the ranges and movement patterns of implanted fish.

Results

Only fish greater than 110 g were used (the majority being greater than 165 g); consequently, transmitters were less than 2.1% of the body weight of all implanted individuals. This size-class represents the largest found in the watershed (McCarthy 1996).

Eighty-eight percent of implanted fish restricted their movements to areas usually less than one-third the size of their home lake. However, some fish in Copper Lake appeared to move throughout very large areas, traveling up to 1 km·day^{-1} within their home lake (Table 1). No fish moved between lakes. Representative ranges exhibited by implanted trout are shown in Figure 2. The majority of fish remained around the shoals near the mouths of tributary streams or along the western side of their home lake. Only three trout moved into tributary streams to spawn. Estimated distances traveled between trackings are recorded in Tables 1 and 2.

Lake spawning was recorded in ten locations; eight within Copper Lake and two within Jim's Lake (Fig. 3). Copper Lake had 47–95 redds within tributary streams and 67–130 redds within the lake. Jim's Lake had 80–250 redds within tributary streams and 55–110 redds within the lake.

On August 28, 1995 one fish (no. 306) was recovered dead in a fyke net at the outflow of Copper Lake. This fish was tagged on August 22, 1995. On September 25, another dead fish (no. 185) was located in a mink (*Mustela visons*) hole about 5 m from the mouth of the tributary stream of Jim's Lake. This fish was implanted on August 18, 1995 and was tracked until its disappearance. One fish from Copper Lake (no. 225) was not located 6 days after being tagged (August 22– 28). This may be a result of transmitter failure or could indicate the fish moved out of the Copper Lake watershed. There are several obstructions (rapids and falls) between Copper Lake and Corner Brook Lake, which is located below the Copper Lake watershed, so regular movement between these two watersheds is not probable.

Table 2. Estimated distance (m) from the previous position for each fish tracked in Jim's Lake, Newfoundland

Date	Transmitter number									
Month/day	025	146	185	205	506	325	345	365	385	405
8/11										
8/15	300				300					
8/16	450				450					
8/17	300	450			300					
8/18	0	450			150					
8/19	150	600	300	0	225					
8/21	150	0	300	0	225					
8/22	0	375	450	0	600					
8/24	0	0	0	0	0					
8/26	450	*[a]	0	0	150	0	150	*	0	150
8/28	375	0	0	0	150	0	75	*	0	150
8/29	75	300	0	0	0	0	0	225	0	600
9/03	0	525	0	0	0	150	0	0	0	600
9/04	150	600	0	0	0	150	0	150	0	0
9/05	75	450	0	0	0	0	0	600	0	0
9/07	150	150	0	0	0	75	0	450	0	0
9/10	150	375	75	0	75	75	75	75	450	0
9/11	450	400	50	0	100	0	25	100	100	0
9/12	100	50	0	0	125	100	0	100	300	0
9/13	300	200	0	0	20	150	0	100	300	100
9/19	350	350	20	0	0	300	0	75	450	100
9/20	0	350	200	50	0	100	50	0	400	50
9/21	0	400	250	50	0	50	100	0	300	50
9/25	50	300	D[b]	0	0	0	0	0	500	0
9/26	150	450		0	0	0	0	0	0	0
10/02	150	250		0	*	0	0	0	50	0
10/04	300	200		0	25	0	0	25	50	0
10/05	30	0		25	25	0	0	0	0	25
10/06	15	400		0	300	25	50	0	15	25
10/07	15	450		0	0	300	300	150	0	25

a * = No signal detected.
b D = Fish was found dead.

Discussion

No trout were re-captured after implantation to check if transmitters interfered with gonad maturation or spawning. Previous studies on the effects of surgical implantation found no significant differences in exhaustion times (Mellas and Haynes 1985), maturation, mortality, or growth of internally implanted salmonids from non-implanted or dummy implanted fish, when transmitters were less than 2% of a fish's total weight (Lucas 1989). All transmitters in this study were less than 2.1% of the implanted fish's total body weight so the effects of implantation were considered minimal.

Of the three fish that moved into the tributary stream of Jim's Lake to spawn, one was implanted on August 24 and the other two were implanted on August 11. All three fish were observed spawning, which also suggested that the transmitters did not impede spawning activity. In addition, two of the three fish were inspected when they went through a counting fence. They were in good condition with closed incisions, lost sutures, and no evidence of infection.

The approximate amount of time trout spent at one spawning location suggested that fish in Copper Lake were much more active during the spawning season than those in Jim's Lake. With trout density in Copper Lake approximately one-third that of Jim's Lake (K.D. Clarke unpublished data), increased movement may have been needed to find other maturing individuals. In addition, all of the implanted fish in Copper Lake may not have been maturing.

The age-at-maturity of the trout populations in each lake showed that 100% of the size-class implanted in Jim's Lake were sexually mature but only 83.3% were mature in Copper Lake (McCarthy 1996). This may also explain why some of the implanted fish in Copper Lake were traveling larger distances as they may have been non-maturing, feeding fish.

This study was intended in part, to determine which of the various tributary streams were preferred spawning habitat; however, it was found that shoals in the lakes are also important spawning habitats within the watershed. Only three of the 16 surviving fish moved into tributary streams to spawn. The others appeared to be associated with lacustrine spawning habitat near the mouths of tributary streams or along the western shores of their home lake. The streamflows during the study were high and access to the streams prior to spawning was not impeded (McCarthy 1996). Visual evidence suggests that these fish were spawning on the shoals.

The western sides of the lakes are characterized by very steep slopes and limited littoral habitat (Scruton et al. 1995). Along these western shores, redds were located on small rock outcrops, which were about 2 m^2. This is an important observation and indicates fish are able to detect and utilize very small and distinct spawning habitats within the lakes.

The amount of lacustrine spawning in Newfoundland may vary based upon the availability of groundwater upwelling (Fraser 1985) and the level of competition for preferred habitat (Cowan and Baggs 1988). In this study, Copper Lake appeared to have a higher percentage of its redds in lacustrine habitat than Jim's Lake. Groundwater upwelling has been strongly associated with brook trout spawning habitat (Fraser 1985; Curry and Noakes 1995; Curry et al. 1995); however, dye dispersion studies over redd sites in ponds on the Avalon Peninsula, Newfoundland, did not reveal any definite aquifer upwellings (Cowan and Baggs 1988). Cowan and Baggs suggested that these redds were produced by brook trout that were displaced from preferred spawning areas. The influence of groundwater on the spawning sites within the Copper Lake watershed was not investigated; therefore, conclusions as to the relative importance of groundwater and competition to the selection of lacustrine spawning sites could not be determined.

In 1995, after a small (1.82 ha) clear-cut was harvested within the drainage basin of the treatment stream (Fig. 1), mean total suspended sediment in that stream rose from 2.0 $mg \cdot L^{-1}$ to 161.1 $mg \cdot L^{-1}$ (McCarthy 1996). This increase in suspended sediment may have led to an increase in the proportion of fish moving out of the stream (McCarthy et al. 1998). Schofield (1993) stated that shoal spawning habitat may be degraded as a result of siltation due to beaver impoundment. Improper forest harvesting, without appropriate no-harvest buffer zones, may also increase siltation (Clarke et al. 1998).

It is important to note that the majority of the lacustrine spawning within Copper Lake seemed to take place on the shoals near the mouths of the tributary streams. These shoals account for about 41% of redds estimated in Copper Lake and could be adversely affected by forest harvesting activities. Shoals are usually beyond the stream itself where streamflows meet the slower water of the lake. This is where sediment carried by the stream would likely be deposited due to a sudden change in water velocity (Swanston 1991). Potential increased sedimentation due to forest harvesting activities, will be assessed in future research.

Conclusions

Lacustrine spawning may represent a large proportion of reproduction in certain areas in Newfoundland and as such need to be considered in the context of effects from forest harvesting practices.

Acknowledgments

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Summary of the Evaluations Following the Forest–Fish Conference, 1996

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Abstract

Following the Forest–Fish Conference, 76 attendees submitted responses to a question asking what critical item they felt should be addressed, and what would be a recommended strategy to mitigate the problem. All responses were reviewed and subsequently placed within one of 20 categories. The management and road crossings accounted for the largest proportion of responses identified by 20.5 and 17% of the number of responses, respectively. The need for training and education, enforcement, and the maintenance of riparian zones/buffer strips accounted for an additional 23.2%, collectively. Respondents identified numerous informative and insightful comments, which have all been presented. The diversity of the identified issues suggests that there are many areas that need to be addressed to lessen the potential impact forest management practices have on the aquatic environment. It is clear that there is a strong need for better exchange of information between all stakeholders.

Monita, D. M. A. 1998. Summary of the evaluations following the forest-fish conference, 1996. Pages 523–530 *in* M.K. Brewin and D.M.A. Monita, tech. coords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1–4, 1996, Calgary, Alberta. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Introduction

During May 1 to 4, 1996, the Forest Fish Conference: Land Management Practices Affecting Aquatic Ecosystems, was held in Calgary, Alberta. The conference had numerous objectives including to facilitate the exchange of information amongst all stakeholders in forest and aquatic resources, to increase awareness of the need for better management solutions, and to increase cooperative working relationships between stakeholders. The conference was well attended by persons from industry, various federal, provincial and state governments, first nations communities, academia, fisheries biologists, consultants, hydrologists, conservation organizations, and concerned individuals.

The conference was generally viewed as overwhelmingly successful, with many of the attendees remarking on its need and timeliness. A wide array of issues concerning the actual and potential effects the forest industry has on aquatic ecosystems were presented during oral and poster presentations. There was plenty of opportunity for candid, informal discussion during and after presentations, breaks, and during a panel discussion.

At the end of the conference, evaluation forms were distributed to attendees. The purpose of the evaluation was to identify what the attendees thought were the most critical questions concerning the potential effects the forest industry has on the aquatic environment. The question on the evaluation was stated as follows:

"This conference has provided a diverse view of the many issues facing the forestry/fish interface. In your view, what is the one critical item which must be addressed? And what would be your recommendation for a strategy to resolve this issue?"

This paper provides a review of how attendees responded to that question.

Methods

A total of 76 evaluations were received. Many of the respondents had identified more that one critical item and recommended strategy, and; consequently, 112 separate issues were identified. Each was placed in one of 20 categories that were chosen based on a review of all the evaluations.

The difficulty in reviewing the responses was that many of the issues border more than one category. For example, some respondents identified the need for research as a critical item, some identified the need for better management practices, and others identified the need for more research to assist in the development of better management practices. A decision was made to place each of the respondent's issues into only one of the 20 categories. Each of the respondent's comments were reviewed and summarized within the appropriate category.

Results

Beside each of the category headings, the number of times an issue was mentioned is put in brackets along with the corresponding percentage (calculated based on the total number of issues [112]). Below each category heading, there is a brief review of the respondent's comments and recommended strategies (if provided). Table 1 provides a summary of categories, the number of times the issue was mentioned and the corresponding percentage.

1. Management (n = 23, 20.5%)

Respondents most frequently identified the need for proper management of the forest resource to minimize the potential impact to the aquatic environment. The development of effective forest management plans, which incorporate values from many interested parties, provide more direction for the forest industry and have flexibility to address site-specific issues were reiterated by the respondents.

The critical issue to many was that all stakeholders with a vested interest in forest and fish resources work cooperatively. Most respondents echoed the call for more integrated management, where there is more direct participation of several groups and individuals who operate with different mandates. Many felt that current guidelines have lacked input from specific groups and a more coordinated approach is needed.

Although there was a lot of good information to make informed decisions, there was a strong need for more technical information that would continue to assist in the formation of more appropriate management decisions. Many expressed dissatisfaction with the fact that management decisions were slow in keeping with new information. Concern had been expressed that the transfer of scientific information should be made in a clear and understandable manner so that the public, the forest industry, and other resource users can understand and implement changes in their decision-making processes. Additionally, many felt that the implementation of decisions must continue; these decisions must be based on the best available science; and these decisions must not be delayed while additional research is conducted.

Table 1. Summary of issues identified by attendees

	Issue	No. times mentioned	Percentage
1.	Management	23	20.5
2.	Road crossings	19	17.0
3.	Training and education	11	9.8
4.	Enforcement	8	7.1
5.	Maintaining buffer strips/Riparian areas	7	6.3
6.	Monitoring	7	6.3
7.	Maintaining natural variability/Diversity	5	4.5
8.	Research	5	4.5
9.	Establishment of partnerships	4	3.6
10.	Standardization between jurisdictions	4	3.6
11.	Holistic approach to solutions	3	2.7
12.	Funding resource management agencies	3	2.7
13.	Angling	2	1.8
14.	Forestry economics	2	1.8
15.	Climatic changes	2	1.8
16.	Limited availability of forest resources	2	1.8
17.	Maintenance of water flow	2	1.8
18.	Nutrient dynamics	1	0.9
19.	Recycling	1	0.9
20.	Inclusion of other industries	1	0.9

The issue of regulations on public versus private land was raised by respondents who suggested that there should be more uniformity. Without a means to address the difference of regulations between public and private lands, it was suggested that regulations for public lands would never achieve the goals of sustainability or biodiversity.

Respondents provided a variety of recommendations to help resolve the issue of a need for better management:

- The sport fisherman, recreation users, scientists, the forest industry, and other stakeholders should join and establish watershed committees that provide a forum to discuss land-use practices that are environmentally sustainable to fisheries habitat.
- A joint body that cooperatively resolves issues of habitat degradation should be established.
- A current need was identified for the Alberta Government to work with stakeholder groups (e.g., industry, NGOs and the public) to improve and develop effective watershed guidelines.
- Industry should be more open to accommodate other resource users to ensure negative impacts to aquatic habitat are avoided.
- Consider involvement of the Center of Excellence for Sustainable Forest Management at the University of Alberta to provide scientific information for to the formation of guidelines.
- Implement the requirement for government and industry to monitor their impacts on watersheds at regular periods.
- Consider an effective regulatory system to ensure watershed guidelines are implemented.
- Develop and ensure the implementation of a watershed assessment methodology prior to road construction.

2. Road Crossings (n = 19, 17.0%)

Many respondents recognized roads used by the forest industry as the major source of sediment to the aquatic ecosystem. Fine coarse particles contained on road surfaces can be transported to the watercourse reducing habitat quality and invoking avoidance responses in fish. Additionally, with the construction of access roads, native vegetation is disturbed, exposing sediments and making them more vulnerable to future transport to the watercourse. The increase of access roads, and the corresponding increase in the probability of sediments being transported into the watercourse, concerned numerous respondents.

Culverts, which typically are used to facilitate water transport beneath roads, were strongly criticized for their prevention of fish migration. Culverts were viewed as physical barriers of migration for reasons that water velocity was often too high

through them and the culverts were frequently overhanging, preventing all fish passage. Culverts were viewed as serious threats to salmonid populations, especially when they were located downstream from spawning or rearing areas. The general feeling was that there are many improvements that could be made to provide better waterway passage beneath roadways.

The respondents presented numerous strategies for reducing the potential adverse affects the construction and use of roads:

- Roads should be constructed, revegetated and reclaimed in a manner that will keep erosion and sedimentation to a minimum.
- Research needs to be conducted into engineering and designing of road crossings that minimize the potential affects on fish and fish habitat.
- The use of fine-grain materials should be avoided during road construction.
- Better information on surface geology should be obtained before setting road density limits.
- Hard packed or paved surfaces should be considered in areas with high erosion potential.
- Alternative log-extraction processes should be considered in areas where the ground is particularly susceptible to erosion.
- After construction of a new road, until the road becomes stabilized, runoff capture techniques need to be implemented.
- Road densities should be kept to a minimum and locate roads away from more sensitive areas.
- Road density and/or crossing density should not be increased if crossing standards on a given system improve.
- More road closures and temporary versus permanent road accesses should be implemented.
- When roads are no longer in use by industry they should be closed to motorized vehicle access to reduce the need for maintenance and to facilitate revegetation and reclamation.
- Forestry roads should be closed permanently after forestry activities are complete.
- There should be more rigid monitoring and enforcement focusing on point sources associated with roads. If regulations are violated, significant penalties should be implemented.
- A workshop should be conducted on how industry and government can work together to address the issues pertaining to road crossings.
- A specific conference should be held on road engineering and its impacts on fish habitat with the objective of devising the optimum management practices to reduce (not mitigate) potential impacts.

3. Training and Education (n = 11, 9.8%)

There were two types of training/education initiatives addressed in the respondents comments: the transfer of knowledge to implement better management practices, and informing the public about the extent and the scope of the existing problem. A strong need for all stakeholders to be involved to help bridge the gap between more divergent viewpoints was identified, and education was seen as a means of improving the current problem.

Some respondents suggested that there appeared to be a lot of knowledge on forestry-related impacts on ecosystems and corresponding operational strategies that could be implemented to mitigate habitat impacts; however, this knowledge needs to be transferred to persons involved in the industry. Several respondents suggested that there was a lack of suitable training and education of those involved directly in the forestry industry (forest managers, logging crews, company officials, etc.). Concern was expressed that the public lacks information on the potential effects of current forestry practices, and without public demand, governments would not show a willingness to enforce current regulations.

There was a strong sense that policy makers, planners, enforcement agencies, forestry companies, public, etc. should all be informed to prevent the 'reinvention of the wheel' and help in the incorporation of more useful practices.

Several of the respondents felt that conferences that address these types of issues (such as this Forestry-Fish conference), workshops, and continuing education courses, should be held on a regular basis to help disseminate new information and keep all parties informed about the latest research, new ideas, etc. It was suggested that it is imperative to keep communication links open to accelerate everyone's level of understanding.

Some respondents suggested that desirable and achievable attributes needed to be defined clearly and guidelines to meet these attributes implemented in terms that everyone in the industry understands. One respondent suggested that in many places there are adequate policies in place but there is inadequate expertise and communication to implement policies.

It was suggested that education among forestry

workers would create more hobby biologists. People in the forest industry must feel that they have the ability and the opportunities to take the necessary conclusions while standing amongst the trees.

One respondent suggested that there was little information, such as that presented at this conference, readily available to the public. This respondent suggested that there was work to be done in the area of developing educational programs and the publication of research results. One respondent suggested that as environmental destruction is taking place at such a rate, that simply working with industry would bring about change too late; there needs to be more public education as it is the public who are most effective in demanding change.

4. Enforcement (n = 8, 7.1%)

In general, respondents were generally concerned that currently existing regulations are too lenient, and there is no effective monitoring for compliance.

While there were not many suggestions on how to resolve this issue, respondents felt that there was insufficient monitoring for compliance of existing regulations, the penalties for non-compliance were too lenient, and the penalties needed to be strengthened. Additionally, there was concern expressed by some respondents about self-monitoring by industry, as it was not being conducted adequately. Some suggested that enforcement needed to be conducted by persons with no vested interest in timber extraction.

One respondent suggested that in the United States timber companies in general have very poor operating records, although they are given charge of all management activities on vast areas of public land. The respondent suggested that there is a better opportunity to improve the standard of compliance in Canada, where there is more interaction between industry and government. However, there must be more willingness and effort of regulators to enforce compliance than has been demonstrated in the past.

Concern was expressed by a number of the respondents that in light of recent budgetary restraints and the consequent reduction in funding for personnel to ensure compliance, standards would diminish. There must be the political will within governments to establish more appropriate legislation and regulations to better protect the natural ecosystem, and secure sufficient funding to ensure compliance.

5. Maintaining Buffer Strips/Riparian Areas (n = 7, 6.3%)

Many issues associated with buffer strips (large woody debris, soil stability, susceptibility of blowdown, etc.) were identified. Although the role and function of riparian areas differs greatly in different habitat types, soil conditions, topography, etc., respondents, in general, felt that the amount of understanding associated with the appropriate sizing of buffer strips was poor. One respondent described the research in this area as chaos. Tremendous variation in buffer strip sizing between provinces and states suggests that little is understood about proper buffer strip sizing.

Most respondents had suggestions on how to help resolve the issue of buffer strip width size:

- Forestry companies should be involved in research programs to determine appropriate sizing of buffer strips.
- Long-term study programs should be undertaken to gather data from the foothills and boreal regions of Alberta.
- Each harvest site needs to be closely evaluated prior to harvest to determine the stability of soil, and harvesters should modify their practices based on the conditions.
- Regulations should be established to protect riparian stands on both public and private lands.
- Under special approval, "environmentally sound management approaches" should be investigated for limited areas.
- Logging on riparian areas should only be permitted on a long-term rotational basis such that a high percentage of the area retains 150-year-old plus trees.
- Research is needed to determine if buffer strips are effective in reducing sedimentation as well as if mitigating measures pursued in the cut blocks are effective in reducing sedimentation.

6. Monitoring (n = 7, 6.3%)

Some of the respondents noted a need for better monitoring of the effects and potential effects of forestry practices on natural ecosystems. Many saw the information collected in monitoring programs as necessary tools that form the basis for making management decisions. Monitoring was also seen as the means in obtaining feedback about the success of existing or past management practices. Monitoring would also provide a larger opportunity for planning and identifying future research priorities.

In general, the respondents' comments suggested a lack of effective monitoring programs to adequately access whether appropriate management practices were implemented. No specific monitoring strategies were identified; however, it was suggested that monitoring both terrestrial and aquatic ecosystems was necessary in order to obtain a better understanding of how both systems change with time, and to get a better understanding of the integration of the two systems. Others suggested that the methods used in the monitoring programs should be customized to the region in which the timber harvests occur.

7. Maintaining Natural Variability / Diversity (n = 5, 4.5%)

The issue of maintaining natural variability in ecosystems was seen as critical to address management decisions. It was suggested that industry needed to be educated on the diversity that exists in ecosystems. Models of variability were proposed as tools for management processes. The maintenance of variability recognizing the occurrence of natural disturbance in aquatic ecosystems should be conducted instead of focusing on homogeneity by concentrating on specific habitat or population goals.

8. Research (n = 5, 4.5%)

Generally, those who identified research as a critical item expressed concern about specific information gaps in the research. Alternatively, one respondent was concerned that current research was geared too much towards specifics, and more information was needed on ecosystem processes. Many suggested that generalities were well understood, but more specific research, which would guide ecologically sound management practices, was needed.

Respondents highlighted the fact that research information must be integrated into management processes. While scientists can't dictate society's direction, they can ensure their work is relevant, directive, and is in a format that is usable by managers.

9. Establishment of Partnerships (n = 4, 3.6%)

Issues related to timber harvesting patterns affect a number of different interested groups including the forestry companies, government, research institutions, the public, etc. Development of effective change to current practices must involve all those concerned with forestry and environmental issues. There should be more dialogue between all stakeholders so that problems and mutual concerns can be worked out. The respondents suggested that more interested groups (company officials, government officials, biologists, and the public) be present at the table when new regulations are developed.

10. Standardization Between Jurisdictions (n = 4, 3.6%)

Some respondents suggested that there should be more standardization of guidelines between different jurisdictions. It was felt that there was a lack of communication between jurisdictions and there may be a lot of 'reinventing the wheel'. It was also recognized that there is tremendous diversity in ecosystems, and what may be issues and potential mitigative measures in one system may not be appropriate in another.

11. Holistic Approach to Solutions (n = 3, 2.7%)

There were a few respondents who felt that often a single species is used for protection purposes, ignoring the complexity and diversity of ecosystems. For example, respondents suggested that there was too much focus only on fish or salmonids in particular. A holistic approach to appropriate management practices would recognize and emphasize the diversity and natural variability in the ecosystem.

12. Funding Resource Management Agencies (n = 3, 2.7%)

Three respondents expressed concern over the lack of funding for agencies charged with the responsibility of resource management. The need for the development and implementation of an effective and broadly supported way to fund the resource management agencies was expressed.

One respondent suggested 3 ways in which resource management agencies could be funded:

1. Dedicated funding (fees paid by industry) directly into a pool from which the resource management agencies budgets are allocated annually.
2. A portion of national sales taxes including the possibility of allowing citizens to dedicate more to fish protection by checking off a voluntary donation box on their tax return.
3. Additional taxes on outdoor equipment.

One respondent was concerned that the apparent conflict that arises when governments who encourage economic growth and job creation are the ones that must develop and enforce regulations.

13. Angling (n = 2, 1.8%)

Two respondents suggested that a critical item was the need to better control angling pressure. Efforts to preserve fisheries by other users of the land (agriculture, oil and gas, etc.) may be for naught in the absence of proper angling management. Increased access to fish by the construction of roads was seen as an issue that needed to be controlled through improved regulations.

14. Forest Economics (n = 2, 1.8%)

Respondents suggested a need for an improved understanding of economic issues facing the forest industry. If people understood the costs associated with environmental protection, perhaps then industry and the public could work together to implement cost-effective, environmental protection measures.

15. Climatic Change (n = 2, 1.8%)

Respondents suggested a need for a better understanding of how forestry activities affect hydrological cycles and weather patterns. Such physical effects to the natural environment could affect maturation rates and migration timing of fish.

16. Limited Availability of Forest Resources (n = 2, 1.8%)

Two respondents suggested that the most critical item was whether enough forested land existed to meet the future demand for timber and still provide flexibility for using a full range of landscape prescriptions necessary to protect fisheries and other resources. It was suggested that with the current trend of resource consumption, mitigative approaches are insufficient, and ultimately, we may be heading towards severe declines of valued native species.

17. Maintenance of Water Flow (n = 2, 1.8%)

Two respondents suggested that water flow and water quality needs to be planned based on the demands by humans for drinking water, fisheries needs, and ecosystem values. Mindful of these values, the amount cut and cutting patterns would change accordingly. The respondents further suggest that the objectives for flow regimes should be set on a watershed basis mindful of a specific time period (i.e., 50 to 100 years). Forestry companies would then plan their timber harvesting practices to incorporate appropriate cutting patterns, buffers, etc. based on these objectives.

18. Nutrient Dynamics (n = 1, 0.9%)

One respondent suggested that there needed to be a better understanding of nutrient dynamics relative to forest harvesting practices.

19. Recycling (n = 1, 0.9%)

One respondent suggested that properly instituted recycling programs would reduce the need for the harvesting of timber.

20. Inclusion of Other Industries (n = 1, 0.9%)

A suggestion was made that potential environmental affects of the oil and gas as well as the agricultural industries needed to be included with discussions about the forest industry.

Discussion

The diversity of issues expressed by the respondents suggest that there are many areas in which there is a need to work on minimizing the effects the forest industry has on the aquatic environment. There are some issues, including the need for exchange of technical information between stakeholders and strong demand for better management practices that are guided by the best available scientific information. The issue of road crossings and their impacts on the aquatic environment was seen as critical for many. One respondent suggested a separate conference be held on this issue alone. Training and education issues were frequently expressed, for not only those involved directly in the industry, but also for the public who are seen as the most effective in demanding change.

Concern was expressed about the lack of sufficient enforcement and monitoring of forestry activities. Many of the additional issues expressed generally identified a need for more information on ecosystem processes, the response of the ecosystem to changes in land use practices, and the communication of this information amongst all stakeholders.

There is new information coming from research, new management ideas and practices, new regulations developed in various jurisdictions, etc., and there needs to be a more effective forum for disseminating this information amongst all stakeholders. The establishment of a more rapid means of communication (list servers, web sites, etc.) addressing these issues is needed.

It is evident there is a strong need for more interaction between stakeholders, and conferences such as this one should be held on a regular basis. It is the

suggestion of the author of this paper that conferences with similar objectives be held on a more frequent basis, perhaps annually. The Forest–Fish Conference was extremely useful, not only in providing a forum for communication, but to identify future needs of all stakeholders. The multi-disciplinary / multi-stakeholder approach to the conference provided a forum for interesting discussions focussing on practical solutions. It provided to its attendees a very useful exchange of viewpoints, ideas and solutions of which, according to responses provided in the evaluations, there is a strong need.

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